BASELINE ECOLOGICAL RISK ASSESSMENT
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

McGinnes Industrial Maintenance Corporation
International Paper Company
U.S. Environmental Protection Agency, Region 6

Prepared by

Integral Consulting Inc.
411 1st Avenue S, Suite 550
Seattle, Washington 98104

August 2012
Revised May 2013

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
95UCL	95 percent upper confidence limit on the mean
AhR	aryl hydrocarbon receptor
AWQC	ambient water quality criteria for the protection of aquatic life
BEHP	bis(2-ethylhexyl)phthalate
BMF	biomagnification factor
bw	body weight
CCC	criterion continuous concentration
CERCLA	Comprehensive Environmental Response, Compensation and Liability Act of 1980
CMC	criterion maximum concentration
COI	chemical of interest
COPC	chemical of potential concern
$COPC_E$	chemical of potential ecological concern
CSM	conceptual site model
CT	central tendency
CTR	critical tissue residue
DMP	Data Management Plan
DQO	Data Quality Objective
dw	dry weight
EcoSSL	ecological soil screening level
EPC	exposure point concentration
ER-L	effects range-low
ER-M	effects range-median
EROD	ethoxyresorufin-*O*-deethylase
FCA	fish collection area
FSR	Field Sampling Report
HQ	hazard quotient
HQ_L	HQ calculated using a lowest-observed-adverse-effects level
HQ_N	HQ calculated using a no-observed-adverse-effects level

I-10	Interstate Highway 10
Integral	Integral Consulting Inc.
IPC	International Paper Company
LOAEL	lowest-observed-adverse-effect level
lw	lipid weight
MIMC	McGinnes Industrial Maintenance Corporation
MWW	Mann Whitney Wilcoxon
NOAEC	no-observed-adverse-effects concentration
NOAEL	no-observed-adverse-effects level
OCDD	octachlorinated dibenzo-*p*-dioxin
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
ppt	parts per thousand
PSCR	Preliminary Site Characterization Report
QA	quality assurance
RACR	Removal Action Completion Report
RBA	relative bioavailability adjustment
REV	reference envelope value
RI/FS	Remedial Investigation and Feasibility Study
RM	reasonable maximum
RMin	reasonable minimum
SAP	Sampling and Analysis Plan
Site	San Jacinto River Waste Pits site in Harris County, Texas
SJRWP	San Jacinto River Waste Pits
SLERA	Screening Level Ecological Risk Assessment
SQG	sediment quality guideline
SSD	species sensitivity distribution
SWAC	surface area-weighted average concentration
TCDD	tetrachlorinated dibenzo-*p*-dioxin
TCDF	tetrachlorinated dibenzofuran
TCEQ	Texas Commission on Environmental Quality

TCRA	time-critical removal action
TEF	toxic equivalency factor
TEQ	toxicity equivalent
TEQ_{DF}	TEQ concentrations calculated using only dioxins and furans
TEQ_{DFP}	TEQ concentrations calculated using dioxins and furans and dioxin-like PCBs
TEQ_{P}	TEQ concentrations calculated using only dioxin-like PCBs are referred to as TEQ_{P}
TMDL	total maximum daily load
TOC	total organic carbon
TRV	toxicity reference value
UAO	Unilateral Administrative Order
UCL	upper confidence limit on the mean
USEPA	U.S. Environmental Protection Agency
ww	wet weight

1 INTRODUCTION

This baseline ecological risk assessment (BERA) was prepared on behalf of International Paper Company (IPC) and McGinnes Industrial Maintenance Corporation (MIMC; collectively referred to as the Respondents) in fulfillment of the 2009 Unilateral Administrative Order (2009 UAO), Docket No. 06-03-10, issued by the U.S. Environmental Protection Agency (USEPA) to IPC and MIMC on November 20, 2009 (USEPA 2009), for the San Jacinto River Waste Pits (SJRWP) site in Harris County, Texas (the Site). The 2009 UAO directs the Respondents to perform a Remedial Investigation and Feasibility Study (RI/FS) for the Site, and indicates that the RI include a BERA. This document fulfills the UAO requirement for the BERA, building on the conceptual site models (CSMs) described in the Preliminary Site Characterization Report (PSCR) for the impoundments north of Interstate Highway 10 (I-10) and surrounding aquatic environments (Figure 1-1).

A Screening Level Ecological Risk Assessment (SLERA) for the overall Site was presented as Appendix B to the RI/FS Work Plan (Anchor QEA and Integral 2010). That SLERA did not address the south impoundment, because it was written prior to USEPA's requirement that the south impoundment undergo investigation. In March 2011, soil samples were collected from the south impoundment area and analyzed for chemicals of interest (COIs). The resulting data have been used to perform a SLERA for the south impoundment, which is included as Appendix E to this document. USEPA has requested that additional studies be conducted with respect to the south impoundment area. This document presents a SLERA for the south impoundment in Appendix E to provide the screening-level problem formulation and the selection of receptors and assessment endpoints. Appendix E also includes analysis of the soil data collected in 2011 and identification of chemicals of potential ecological concern ($COPC_{ES}$) for ecological receptors that may use that area. Following USEPA approval of this draft south impoundment SLERA and completion of the investigation of that part of the Site, a BERA for the south impoundment will be prepared. It will be presented in the Remedial Investigation Report.

1.1 Purpose

The Comprehensive Environmental Response, Compensation and Liability Act of 1980 (CERCLA) guidance requires that remedies at contaminated sites be protective of human health and the environment (USEPA 1988). The baseline risk assessments evaluate the potential threats to human health and the environment in the absence of any remedial action, help determine whether remedial action is needed, and serve as the basis for the evaluation of the effectiveness of any subsequent remedial action. Ecological risk assessment addresses the likelihood that adverse effects on the environment, and to specific ecological receptors, may occur or are occurring as a result of exposure to one or more stressors (USEPA 1997).

The purpose of this BERA is to determine the nature and magnitude of risks to ecological receptors that result from any releases of hazardous substances from the impoundments north of I-10 at the Site. Results of the baseline risk assessments support risk managers by providing a point of reference for evaluation of the no-action alternative in the feasibility study, and for quantification of risk reduction that can be achieved by each remedial alternative considered in the feasibility study.

1.2 Document Organization

The approaches and methodologies presented in this BERA are consistent with USEPA guidance for conducting ecological risk assessments (USEPA 1997, 1998), and with Data Quality Objectives (DQOs) and related statements and information presented by the sediment, tissue, and soil sampling and analysis plans (SAPs) (Integral and Anchor QEA 2010; Integral 2010a, 2011a), and the RI/FS Work Plan (Anchor QEA and Integral 2010a). The document is organized according to specifications in the *Guidelines for Ecological Risk Assessment* (USEPA 1998), and includes the following:

- Section 2. Background Information
- Section 3. Problem Formulation
- Section 4. Exposure Assessment
- Section 5. Effects Characterization
- Section 6. Risk Characterization
- Section 7. Uncertainty Analysis
- Section 8. Summary of Ecological Risks and Risk Conclusions.

This document also includes six Appendices:

- Appendix A Receptor Profiles
- Appendix B Ecotoxicity Profiles
- Appendix C Exposure Point Concentrations Used for Exposure Assessment in the BERA
- Appendix D Estimation of Dioxin and Furan Concentrations in Terrestrial Invertebrate Tissue for the Exposure Model
- Appendix E Screening-Level Ecological Risk Assessment, South Impoundment
- Appendix F EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses.

2 BACKGROUND INFORMATION

This BERA is presented to USEPA following completion of several studies and documents and provides a key component of the analyses required for the RI Report. Relevant background information on the Site setting and CSMs, and information supporting determination of the baseline dataset have been described previously. This section briefly reviews information relevant to the BERA that has been presented in earlier, approved documents. The problem formulation is presented subsequently in this context.

2.1 Site Setting and General Conceptual Site Models

The Site setting was described in the RI/FS Work Plan (Anchor QEA and Integral 2010) and later updated in the PSCR (Integral and Anchor QEA 2012). The PSCR provides a detailed description of the topography, hydrology, hydrogeology, and hydrodynamic environment at the Site. The draft Chemical Fate and Transport Modeling Report (Anchor QEA 2012) provides additional detail on the hydrodynamics and sediment physical environment, as well the fate of 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxin (TCDD), 2,3,7,8-tetrachlorinated dibenzofuran (TCDF), and octachlorinated dibenzo-*p*-dioxin (OCDD).

Also described in the PSCR are two CSMs that provide the basis for the ecological exposure and risk analyses. These most recent iterations of the CSMs form the conceptual framework of chemical transport and exposure pathways that could lead to exposure of ecological receptors. Existing CSMs describe the environment of the northern and southern impoundments in the following general context:

- **The area north of I-10 and surrounding aquatic environment.** This area consists of a set of impoundments approximately 14 acres in size, built in the mid-1960s for disposal of paper mill wastes, and the surrounding areas containing sediments and soils potentially contaminated with chemicals originating in the waste materials that had been disposed of in the impoundments. The set of impoundments is located on a partially submerged 20-acre parcel on the western bank of the San Jacinto River, immediately north of the I-10 bridge (Figure 2-1). Dredging activities by third parties have occurred in the vicinity of the perimeter berm at the northwest corner of these impoundments; samples of sediment in nearby waters north and west of these impoundments indicate that dioxins and furans are present in nearby sediments.

Other sources of dioxins and furans are present upstream and on the Site, including chemical manufacturing facility outfalls, wastewater treatment plant outfalls, stormwater runoff and outfalls, and atmospheric deposition (Integral and Anchor QEA 2012). The Baytown West Wastewater Treatment Plan outfall occurs directly north of the I-10 bridge on the river's eastern shore. University of Houston and Parsons (2006) presents information on dioxins and furans in effluent from this wastewater treatment plant. Also on the eastern shore and to the north is a stormwater outfall draining a very large area, and atmospheric sources of dioxins and furans are present on the Site as well. Section 4.2.1 of the PSCR provides additional detail. The CSM that provides a summary of the chemical sources and the release and transport pathways is depicted in Figure 1-1.

- **The impoundment south of I-10.** Another impoundment may be present south of I–10, on the peninsula of land south of the 20-acre parcel. Portions of the peninsula are believed to have been used in the 1960s as a disposal area for paper mill waste similar to that disposed of in the impoundments north of I-10. Currently available information about the area south of I-10 indicates that wastes other than those originating from the Champion Papers Inc. paper mill were also deposited in the impoundment (Integral and Anchor QEA 2012), but the origins of the other waste and debris in that area are unknown. The CSM for the south impoundment primarily addresses the terrestrial environment (Figure 2-2); USEPA has requested additional studies to address data gaps and identify materials present in the impoundments south of I-10.

Finally, since this Site was added to the National Priorities List in 2008, a time-critical removal action (TCRA) has been implemented. Construction of the TCRA, which involved installation of a cap over the area within the original perimeter of the impoundments north of I-10, was completed in July 2011. The TCRA is relevant to the BERA because it has substantially changed ecological conditions and exposure pathways at the Site (Figure 2-3), reducing the potential for exposure of ecological receptors to the contaminated waste and sediment present on the Site. Details describing implementation of the TCRA can be found in the draft Removal Action Completion Report (RACR) (Anchor QEA 2011a).

Problem formulation (Section 3) integrates available information to describe specific pathways and exposure routes of interest to ecological receptors in the area north of I-10 and aquatic environment, building from the CSM of that area described above. The CSM describing ecological exposures, and from which the analysis steps in the BERA are determined, are detailed in Section 3.8. The south impoundment SLERA, including a screening level problem formulation and a CSM for ecological receptors, is presented in Appendix E.

2.2 Baseline Risk Assessment Datasets and Data Treatment Rules

Determination of an appropriate baseline dataset, which will be used to describe the current site conditions, is a key step of the RI/FS process. Once the appropriate data are identified, calculations are performed using a specified set of data treatment rules.

2.2.1 Baseline Dataset

The RI/FS Work Plan describes the rationale for selection of data to be used in the baseline risk assessments: data to be used in the baseline risk assessments should be of known quality, which includes only Category 1 data (as described in Section 3 of the RI/FS Work Plan) and should reflect the current, pre-remediation condition, which does not include conditions present in 2005 or previously (Integral 2011c). The Exposure Assessment Memorandum (Integral 2012) describes the process for incorporating additional data for polychlorinated biphenyl (PCB) congeners in catfish fillet and sediment collected on Site by the Texas Commission on Environmental Quality's (TCEQ) Total Maximum Daily Load (TMDL) program in 2008 and 2009 (University of Houston and Parsons 2009; Koenig 2010, Pers. Comm.). Appendix A to the Exposure Assessment Memorandum (Integral 2012) documents Integral Consulting Inc.'s (Integral's) independent validation of TCEQ's PCB congener data for tissue and sediment on the Site and for tissues in background areas according to procedures described by the RI/FS Work Plan. This validation effort resulted in a change to the classification of these PCB data from Category 2 to Category 1.

Both Site and background data are used in the risk assessment. Analysis of background information allows for consideration of other sources of risk at the Site, which is relevant to both risk assessment and evaluation of remedial alternatives. Background conditions provide the basis for understanding the incremental risks due to the site (Section 3.8.4.5). Such context informs risk management by ensuring that remedial actions that may be taken at the Site will actually result in reduction of exposure and risk originating from Site-related sources.

The baseline dataset for the Site consists of:

- Sediment, tissue, and soil data collected for the RI/FS (all new data collected by respondents since December 2009), including soil from the south impoundment collected in 2012[1]
- Sediment and surface water data collected by URS (2010) for TCEQ in 2009.
- PCB congener data for fish tissue and sediments resulting from sampling conducted by TCEQ in 2008 and 2009.

The background dataset developed for the RI consists of:

- Upstream surface (0- to 6-inch) sediment samples for 21 subtidal and 10 intertidal locations
- Background soil samples (0- to 6-inch and 6- to 12-inch) from 10 locations in the I-10 Beltway 8 East Green Space and from 10 locations in Burnet Park
- Clam and killifish from two locations upstream and hardhead catfish and blue crab from locations in Cedar Bayou.[2]

Background tissue and soil data were collected prior to publication of the PSCR (Integral and Anchor QEA 2012) and are described in that document; additional sediment samples were collected upstream in 2011 and will be described in the RI Report.

Although the ecological risk assessment uses different data types and uses data differently than the human health risk assessment, the baseline dataset described above is comprehensive for both of the baseline risk assessments to be conducted for the RI.

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[1] Sampling is documented in Addendum 3 to the Soil SAP for additional soil sampling south of I-10 (Integral 2011d); Addendum 2 to the Sediment SAP (Integral 2011b) and Addendum 1 to the Groundwater SAP (Anchor QEA 2011b).

[2] Background tissue data have also been collected for edible crab and catfish south of the Fred Hartman bridge; these data are for use in the human health risk assessment only.

2.2.2 Data Treatment Rules

RI/FS data are managed according to the project Data Management Plan (DMP), which is provided as Appendix A to the RI/FS Work Plan. Section 6.5 of the DMP also describes data averaging rules such as the averaging of results for replicates and treatment of qualified data. Data accessed for analyses in this report were prepared according to those rules. For performance of various analyses in this report, general data treatment rules are as follows:

- Nondetects were estimated at one-half the detection limit for use in all calculations, unless otherwise specified.
- TCDD toxicity equivalent (TEQ) concentrations were calculated using the toxicity equivalency factors (TEFs) most appropriate for the receptor being analyzed. These are discussed further in Section 3.2.
- TEQ concentrations in samples for which one or more dioxin and furan congeners were not detected were calculated using nondetects equal to one-half the detection limit. TEQ concentrations for PCB congeners for which one or more PCB congeners was not detected were calculated using nondetects equal to one-half the detection limit. If one or more congener concentration was estimated in calculation of a TEQ concentration, the TEQ is reported as estimated (J-qualified) in the database. If all congeners were not detected in a sample, the TEQ is reported as not detected (U–qualified).
- Any nondetects for a given analyte and medium that were higher than the maximum detected concentration for the same analyte and medium were considered "high-biasing non-detects," and were removed prior to use of the dataset in the BERA, following USEPA (1989) guidance.

In the calculation of exposure point concentrations (EPCs), and in statistical evaluations of the datasets (e.g., characterization of data distributions), specific rules were applied for estimating values for the censored data. Procedures for substituting values for censored data varied, depending on the sample size and the detection frequency, as follows:

- For each dataset used in calculation of an EPC or in evaluating the data distribution, the detection frequency was calculated as the percentage of values flagged with a "U" qualifier (not detected).

- Nondetects in datasets with sample sizes equal to or greater than 10 and detection frequencies equal to or greater than 50 percent were set to one-half the detection limit and included in all calculations.
- Datasets with sample sizes equal to or greater than 10 and detection frequencies between 20 and 50 percent were addressed using statistical substitution methods. The substitution method used depends on the distribution of the dataset; for normally or lognormally distributed data, upper confidence limits on the mean (UCLs) were estimated using robust regression on order statistics (Helsel 2005); for datasets with unknown data distributions (those that could not be defined as normal or lognormal), a nonparametric Kaplan Meier approach for imputing nondetects was used (Helsel 2005; Singh et al. 2006).
- Nondetects in datasets with sample sizes less than 10, regardless of detection frequency, or in datasets with detection frequencies less than 20 percent, regardless of sample size, were not subject to statistically derived substitutions, because the pool from which information about the data distribution can be drawn is insufficient for robust substitution methods. These datasets were treated with nondetects substituted at one-half the detection limit.

Finally, the data to describe PCBs in the media sampled on Site is variable. In sediment, dioxin-like PCBs were measured in a subset of the samples collected to describe nature and extent of contamination. Within the northern impoundments, samples were collected and analyzed for Aroclors, and elevated detection limits resulted from matrix interferences in several samples from the western cell. In soils, either PCB congener or Aroclor data are available, and in some samples, PCBs were not analyzed. Finally, in tissue, all 209 PCB congeners were measured in all samples. Data treatment rules for calculation of aggregate variables for PCBs (total PCBs and TEQ_P) were consistent with those laid out above. For total PCBs as a sum of Aroclors, this approach is likely to overestimate total PCB concentrations because of the inflated detection limits in some sediment samples.

3 PROBLEM FORMULATION

The problem formulation for the BERA provides a synthesis of ecological conditions, information on fate and transport and relevant toxicological information available at the start of the risk assessment process to finalize the assessment endpoints and risk questions to be addressed by the BERA (USEPA 1998). Information contained in or generated by earlier SJRWP documents prepared under the RI/FS is assembled in this section to provide a complete problem formulation, which results in definition of the approaches and methods used to perform the BERA. Specifically, this section draws on the following previously approved documents:

- The RI/FS Work Plan (Anchor QEA and Integral 2010), which describes the methods and approach to be used to perform the BERA, and provides the SLERA for the area north of I-10 and aquatic environment as an appendix. The SLERA identifies the species potentially present at the Site, including threatened and endangered species, specifies ecological receptors to serve as surrogates or representatives for the species potentially present, and provides a preliminary summary of ecotoxicological information for dioxins and furans.
- The Sediment SAP and the COPC [Chemical of Potential Concern] Technical Memorandum (Integral 2011c), both of which address selection of COPCES. Components of the Sediment SAP addressing selection of COPCs were excerpted and included as Appendix C of the RI/FS Work Plan.
- Technical Memorandum on Bioaccumulation Modeling (Integral 2010b), which addresses patterns of dioxin and furan bioaccumulation in invertebrates, fish, and birds using scientific literature and analyses of site and regional data. This technical memorandum includes results of statistical analyses that may be used to predict tissue concentrations of dioxins and furans from sediment or water concentrations, and an analysis of dioxin and furan bioaccumulation patterns in the area surrounding the Site.
- The PSCR (Integral and Anchor QEA 2012) for the most recent iteration of the CSMs. The latest CSMs were developed from the initial CSMs developed in the RI/FS Work Plan and Soil SAP Addendum 1. Although CSMs are introduced above, the problem formulation describes the final evaluation of exposure pathways relevant to the ecological risk assessment.

This BERA incorporates information from the entire area within USEPA's preliminary Site perimeter, and information from all of these documents as well as other publications. Dioxins and furans were identified in Appendix C of the RI/FS Work Plan as indicator chemicals for the purposes of the remedial investigation. This designation acknowledges that the relatively high toxicity of dioxins and furans in combination with the relatively elevated concentrations make dioxins and furans the focus of risk evaluation and risk management. For this reason, this BERA often incorporates more detail and depth for the evaluation of risks due to dioxins and furans, while using a generally conservative approach to address risks from the other COPCEs.

3.1 Chemicals of Potential Ecological Concern

COPCEs for the area north of I-10 and surrounding aquatic environments (Table 3-1) were determined using the RI dataset according to methods specified in the COPC Technical Memorandum, and are subdivided into those of potential concern to benthic invertebrates and those of concern to fish and wildlife. Chemicals in sediment with a detection frequency of at least 5 percent (following the sediment study) that were either a) present in at least one sample at a concentration greater than sediment screening concentrations protective of benthic invertebrate communities, or b) have no screening value protective of benthic invertebrate communities and c) were not correlated with dioxins and furans, are considered COPCEs for benthic macroinvertebrate communities. If a chemical was detected in greater than 5 percent of sediment samples in the RI dataset, and is thought to be bioaccumulative (TCEQ 2006), it is considered to be a COPCE to be evaluated for fish and wildlife.

3.2 Overview of Ecological Effects

All of the COPCEs have some potential to adversely affect the survival, growth, and/or reproduction of one or more ecological receptors if exposures are sufficiently elevated. Information about the types of effects associated with each COPCE in various species, and the information used to interpret exposure estimates for ecological receptors in the BERA are provided in Appendix B.

As the indicator chemical group, dioxins and furans are expected to be the most important ecological risk driver at the Site. In this context, the following section explains how the toxicity of dioxins, furans, and "dioxin-like" PCBs are assessed in this BERA, and provides an overview of the potential biological and ecological effects of all of the dioxin-like compounds in broad categories of ecological receptors. The approach used in this document is consistent with USEPA's (2008) Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. Much of this information has been presented in prior submittals (Appendix B to the RI/FS Work Plan; Exposure Assessment Memorandum). More detailed discussion of the toxicity reference values and benchmarks used to interpret exposure estimates for dioxins and furans as well as the other COPCES is provided in Appendix B of this document.

3.2.1 Evaluating Exposure and Toxicity of "Dioxin-Like" Compounds

For each of 17 dioxin and furan congeners with chlorine substitutions in the 2,3,7,8-positions of the molecule, toxicity to fish, birds, and mammals is widely believed to occur through a common biochemical mechanism, one that is initiated by the binding of the congener to the aryl hydrocarbon receptor (AhR). Of the 17 AhR-active congeners, 2,3,7,8-TCDD exhibits the greatest potential for binding with AhR in many assays. The common toxicological mechanism among the 17 congeners, and the availability of a single potency index (2,3,7,8-TCDD potency) provides the basis for calculating the cumulative exposure to all AhR-active congeners for the purposes of evaluating toxicity and establishing thresholds of toxicological effects. The magnitude of toxicity of each of these 17 dioxin and furan congeners can be related to the toxicity of 2,3,7,8-TCDD using a congener-specific TEF. The concentration of each congener is converted to equivalent concentrations of 2,3,7,8-TCDD by multiplication with its TEF, and all the TEQs for individual congeners (the product of each congener and its TEF) are added to compute the total toxic equivalency of the mixture to 2,3,7,8-TCDD. The resulting total TEQ concentration provides the metric of exposure to "dioxin-like" compounds. Separate sets of TEFs have been derived for mammals, birds, and fish, and are provided in Table 3-2.

The toxic equivalency approach was first developed in 1977 for screening risks from dioxins and furans in combustion sources and incinerator emissions (Eadon et al. 1986; Erickson

1997). It was first used as an "interim" screening tool to evaluate the toxicity of mixtures of dioxins and furans. In 1989, USEPA stated that the TEQ approach "remains 'interim' in character and should be replaced as soon as practicable with a bioassay method" (USEPA 1989). The toxicological basis and rationale for the use of the TEF approach is described in Van den Berg et al. (1998; 2006). Guidance for the use of this methodology in ecological risk assessment is provided by USEPA (2008). This guidance explicitly recognizes that, due to interspecies variability in biochemistry and sensitivity to dioxin toxicity, the TEFs in Table 3-2 may be substituted with species-specific TEF schemes in ecological risk assessment, if sufficient rationale is provided. Although this document does not propose alternative TEF schemes, USEPA (2008) guidance highlights the uncertainty of using the TEF methodology across a wide range of species.

The application of the TEQ approach to PCB congeners was introduced in 1991. For some species and some types of toxicological endpoints, 12 of the 209 PCB congeners are considered to have dioxin-like toxicity, and as a result, these PCB congeners are considered to be additive with TEQs calculated from dioxins and furans (Safe 1990). TEFs for these 12 PCB congeners were assigned on the basis of a variety of endpoints demonstrated by *in vitro* assays and *in vivo* animal studies, most of which are noncancer endpoints (Van den Berg et al. 1998). Concentrations of the dioxin-like PCB congeners within a PCB mixture are first converted to TEQ concentrations using various TEFs. Once the TEQs have been calculated for each dioxin-like congener, they can be added to TEQs for dioxin and furan congeners to determine a total TEQ concentration.

3.2.2 TEQ Nomenclature

Toxicity equivalents are calculated and presented in several different ways for ecological risk assessment. To simplify presentation of these concepts, the term "TEQ" is qualified using subscripts to indicate the congeners included in its calculation, and the TEF scheme applied.

- TEQ concentrations calculated using only dioxins and furans are referred to as TEQ_{DF}
- TEQ concentrations calculated using only dioxin-like PCBs are referred to as TEQ_{P}
- TEQ concentrations calculated using dioxins and furans and dioxin-like PCBs are referred to as TEQ_{DFP}.

To specify the TEF scheme used, an additional subscript is applied, including “F” for the use of TEFs for fish; “M” for the use of TEFs for mammals; and “B” for the use of TEFs for birds. For example, using this notation scheme, the following indicates a TEQ calculated for dioxins, furans, and dioxin-like PCBs using TEFs for fish: $TEQ_{DFP,F}$. If the term “TEQ” is used with no subscript notation, it is to reference a general concept and not a specific concentration. TEFs for fish and birds are taken from Van den Berg et al. (1998); TEFs for mammals are taken from Van den Berg et al. (2006).

3.2.3 Mechanisms of Toxicity

In vertebrates, interactions of 2,3,7,8-TCDD and the dioxin-like compounds with AhR leads to alterations in gene expression and signal transduction that are believed to be the biochemical determinants of toxic effects (Birnbaum 1994). AhR is a member of a family of transcription factors that includes aryl hydrocarbon receptor nuclear translocators (e.g., ARNT, ARNT2, and ARNT3) and others. These proteins are involved in the sensation of and adaptation to changing environmental and developmental conditions. Once activated, AhR combines with ARNT and moves into the cell nucleus, where the complex can then bind specific DNA sequences, leading to altered gene expression. A role of the ligand-AhR complex in non-nuclear signal transduction has also been proposed. The functional consequences of AhR activation in fish and wildlife are diverse and involve numerous target organs, including the liver, thyroid, heart, immune system, and reproductive system (Fox 2001; Carney et al. 2006). Although AhR homologues have been identified in various invertebrate species, invertebrate AhR homologues lack specific, high-affinity binding for TCDD and other prototypical AhR ligands (Hahn et al. 1992; Butler et al. 2001).

There is also potential for non-AhR-mediated dioxin and furan toxicity in both vertebrates and invertebrates, but at much higher doses (USEPA 2008). Non-AhR-mediated dioxin and furan toxicity to vertebrates is not addressed because AhR-mediated toxicity is expected to occur at lower exposures. For invertebrates, non-AhR-mediated toxicity is addressed where data are available.

3.2.4 Overview of Toxicity to Ecological Receptors

From an ecological risk perspective, adverse effects of dioxins and furans on reproductive success, growth, and survival are relevant to evaluating the potential for population-level effects in any receptor. A range of reproductive and developmental effects such as reduced fertility, early-stage embryotoxicity, early life-stage mortality, and reduced growth and development of offspring are also relevant reproductive effects, because these effects can conceivably affect the growth or viability of a population. Studies have shown substantial inter-species and inter-taxa differences in susceptibility to the adverse effects of dioxins and furans, which presents a challenge for interpreting risks to species that have not been tested.

Below is a summary of information on the toxicity of dioxin-like compounds to broad categories of ecological receptors. This information was presented in greater detail in Appendix B of the RI/FS Work Plan (Anchor QEA and Integral 2010). Detailed information on the potential effects of $COPC_{ES}$ needed to support this risk assessment is presented in Appendix B.

3.2.4.1 Benthic Macroinvertebrates

The literature on the toxicity of dioxins and furans to aquatic invertebrates is less extensive than for fish, birds, and mammals, and the majority of studies were published more than a decade ago (USEPA 2008; Anchor QEA and Integral 2010). Most of these historical studies have found that aquatic invertebrates are relatively insensitive to dioxin toxicity. Studies summarized in Appendix B include tests on crustaceans, molluscs, insects, oligochaetes, and polychaetes from freshwater, estuarine, and marine environments. Exposure routes tested among these studies include ingestion and direct exposure to contaminated water and contaminated sediment, and several studies note whole body concentrations in animals exhibiting no adverse responses. While the full spectra of possible exposures and invertebrate taxa are not represented by the data, the evidence generally indicates that invertebrates can tolerate relatively high exposures to TCDD, and in some cases, to TCDF as well. Other 2,3,7,8-substituted congeners are not as well studied.

Recent studies have found that bivalve molluscs exhibit reproductive and developmental effects in response to exposure to 2,3,7,8-TCDD (Cooper and Wintermyer 2009) at exposures lower than no-effects levels for other tested species. The mechanism by which 2,3,7,8-TCDD affects bivalve molluscs has yet to be identified with certainty, but researchers agree that it is independent of AhR homologues (Cooper and Wintermyer 2009; Wintermyer and Cooper 2007; Butler et al. 2004). It is possible that other kinds of invertebrates may exhibit reproductive and developmental effects following exposure to 2,3,7,8-TCDD, or other dioxins and furans; most historical studies have evaluated only survival or growth in adult organisms, following exposures only to 2,3,7,8-TCDD. The results of these studies with bivalves will be used to interpret Site-specific invertebrate tissue data.

3.2.4.2 Fish

Substantial literature indicates that dioxin toxicity is mediated through AhR in fish, with some species having more than one receptor type of varying functionality. The period of greatest sensitivity to dioxins of many fish species is the embryo and post-hatch stages, with toxicity manifesting at the lower exposure concentrations as edema, with circulatory and metabolic changes leading to secondary effects. Cartilaginous malformations and growth effects also occur but may be less sensitive endpoints than edema and cardiac effects. Reproductive effects have been shown to occur in the range of thousands of nanograms per kilogram of tissue. Sublethal effects on juveniles and adults are less well studied; however, the literature reviewed suggests that these later life stages are not as sensitive to dioxin toxicity as are early life stages. The literature suggests that population resistance to dioxin toxicity can also occur over time in some fish, as shown for a killifish population living in the vicinity of a Superfund site with high dioxin levels (Nacci et al. 2002).

Within species, many dose response curves are steep, reflecting a relatively narrow range within which toxicity can manifest. However, there is a high level of variability in sensitivity to dioxins among fish species, with effects associated with concentrations ranging over two orders of magnitude. This variability argues for the development of a species sensitivity distribution (SSD) for evaluating effect levels relevant for risk assessment. Expression of dioxin and furan exposures as concentrations in egg tissue (a tissue residue-based effect level) is an appropriate basis on which to express exposure. Effects in early life stage fish appear to be relatively independent of the route of exposure (Steevens et al. 2005) such that studies using a variety of methods of egg exposure (e.g., water, egg injection) are appropriate for effects evaluation. Steevens et al. (2005) derived two SSDs providing excellent representation of the early life stage toxicity of 2,3,7,8-TCDD to fish.

Population-level risks may be mediated by interspecies and interpopulation differences in sensitivity to dioxins, as well as differential sensitivity to different types of dioxins and furans. Sensitivity to dioxins is not necessarily static within species; some fish populations with long-term exposure to contaminated conditions have apparently developed a resistance to toxicity of dioxins and PCBs. At a Superfund site in New Bedford Harbor, Massachusetts, with high concentrations of dioxins, PCBs, and other hazardous chemicals, the resident *Fundulus heteroclitus* (killifish) population has been shown to exhibit heritable resistance to PCBs in terms of reduced mortality relative to populations from uncontaminated reference sites (Nacci et al. 2002).

3.2.4.3 Reptiles

Generally, the literature describing potential effects of environmental toxicants to reptiles is poor, and available data are dominated by studies on turtles, with a paucity of information on lizards and snakes (Sparling et al. 2000b). Portelli and Bishop (2000) performed a review of the literature for organic chemicals other than pesticides. They found no reports of reptiles dying as a result of PCB, dioxin, or furan exposure, despite fairly elevated concentrations in tissues of specimens captured in the wild. Bishop et al. (1991) reported developmental abnormalities (e.g., abnormal eyes, claws and bills) and behavioral abnormalities in turtles exposed to dioxins, furans, PCBs, and organochlorine pesticides, but dose-response relationships have not been reported. This and other studies cited by Portelli and Bishop (2000) suggest correlations between concentrations of PCBs, polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofuran (PCDFs) and abnormalities in developing embryos, but these data are confounded by the presence of pesticides and other chemicals in the environment and tissues of organisms studied.

One recent laboratory study (Hecker et al. 2006) induced ethoxyresorufin-*O*-deethylase (EROD) activity in hepatocytes from the African brown snake (*Lamprophis fuliginosus*) by *in vitro* exposure to TCDD and PCB126, but dose-dependent EROD activity was not induced by two other dioxin-like PCB congeners. Portelli and Bishop (2000) note that there is no correlation between dioxin, furan, and PCBs in eggs and incidence of abnormalities when TEQ was used to characterize exposure, regardless of the TEF scheme used. More information is needed to understand the extent of both potential AhR-mediated toxicity and other toxicity of dioxins and dioxin-like compounds in reptiles.

3.2.4.4 Birds

Exposure to dioxins and furans is associated with adverse effects on bird reproduction and development. Changes in the heart have also been observed, although in some cases the impact of these effects on ultimate reproductive success or survival is unclear. Early life stages, including the embryo and recently hatched chick, appear to be the most sensitive to dioxin toxicity, and an overview of the literature appears to confirm USEPA's (2003b, 2008) position that egg exposure is an appropriate basis for predicting effects. The literature indicates there are substantial differences in susceptibility and sensitivity among species, even within the limits of reproductive toxicity. Among tested bird species, sensitivity to early life stage toxicity spans several orders of magnitude. As for fish (above), the similarity in the ranges of the sublethal and lethal effect concentrations reflects the steep dose-response associated with dioxin-like toxicity.

3.2.4.5 Mammals

Exposure to dioxins and furans is associated with adverse effects on mammalian reproduction and development, more so for the rat than for the mink. Early life stages, including the fetus and newly born pup/kit, appear to be the most sensitive to dioxin toxicity. Similar to birds, there is substantial inter-species variability in sensitivity to dioxins and furans.

Concentrations of dioxins and furans are commonly measured in liver or adipose tissue, because lipophilic compounds such as TCDD may accumulate in these tissues. Moreover, due to toxicokinetic differences between species, administered dose or content of compounds in foods is not as reliable an indicator of exposure to dioxins and furans as organ concentrations. Whole-body or tissue burden is the preferred metric of exposure in laboratory studies and may facilitate inter-species and inter-study comparisons. However, reliable literature on mink expresses exposure as ingested dose (e.g., Zwiernik et al. 2009), providing a useful, non-invasive means of evaluating exposure-response in a risk assessment context.

3.3 Fate and Transport of Chemicals of Potential Ecological Concern

Fate and transport processes relevant to the BERA include uptake and bioaccumulation, biomagnification, degradation and weathering of compounds, and the sequestration of chemicals in environmental matrices such as soils or sediments. The term "bioaccumulation" describes the extent to which an organism retains substances following uptake through any exposure route, resulting in the organism having a higher concentration in its tissues than in the surrounding environment (USGS 2007). "Biomagnification" occurs if concentrations are increasingly greater in higher trophic levels (USGS Toxic Substances Hydrology Program).

For the BERA, Site-specific data describing concentrations of $COPC_{ES}$ in tissues of some organisms are used to estimate the ingestion rate of $COPC_{ES}$ by birds and mammals, and in some cases, are directly compared to toxicity reference values (TRVs) to evaluate the potential for effects. With the exception of the Technical Memorandum on Bioaccumulation Modeling, which reviews patterns of dioxin and furan bioaccumulation in aquatic macroinvertebrates, fish and birds, bioaccumulation has been addressed by following TCEQ guidance; specifically TCEQ's list of chemicals considered bioaccumulative (TCEQ 2006). This list of bioaccumulative chemicals was specifically consulted in selection of COPCs for fish, reptiles, birds, and mammals, because these receptor groups are likely to be significantly exposed to bioaccumulative contaminants through ingestion of prey. Chemicals that are $COPC_{ES}$ (in addition to dioxins and furans) and listed by TCEQ as bioaccumulative from sediments include PCBs, cadmium, copper, mercury, nickel, and zinc. Site-specific data for fish, clam, and crab tissues provide empirical evidence of the bioaccumulation potential of each chemical, and are used as appropriate to evaluate species-specific exposures in the BERA.

The Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) uses site specific data, regional data, and the literature to describe controls on bioaccumulation of dioxins and furans, and resulting bioaccumulation patterns. The technical memorandum finds several lines of evidence indicating that 1) rates of uptake of dioxin and furan congeners by both vertebrates and invertebrates for which data are available are variable (e.g., Tietge et al. 1998), and are controlled to a large extent by the size of the molecule, with the smaller, lower-chlorinated congeners taken up more readily across gill and gut membranes than the larger, more chlorinated congeners (Opperhuizen and Sijm 1990); 2) dioxins and furans can be metabolized and excreted, and this also occurs at different rates for different congeners (Hu and Bunce 1999; Nichols et al. 1998); 3) metabolism results in generation of soluble moieties which can be excreted, and does not occur by dehalogenation, except in bacteria (Hu and Bunce 1999); 4) elimination rates of tetrachlorinated congeners are lower than the more chlorinated congeners (e.g., Niimi 1996); and 5) dioxins and furans do not biomagnify, unlike PCBs which do biomagnify (Naito et al. 2003; Wan et al. 2005; Broman et al. 1992; and Jarman et al. 1997).

The Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) also reports Site-specific statistical regression models that can be used to predict tissue concentrations for some congeners with a measurable degree of uncertainty. Both the conceptual model of bioaccumulation reported by the technical memorandum, and the regression models reported are used in the baseline risk assessments. Analysis of Site-specific tissue data in the PSCR (Integral and Anchor QEA 2012) supported the conceptual model of bioaccumulation developed by Integral (2010b).

3.4 Ecosystems Potentially at Risk

The Site is located in a low gradient, tidal estuary near the confluence of the San Jacinto River and the Houston Ship Channel. The surrounding area includes a mix of land uses, including two constructed reservoirs: Lynchburg Reservoir to the southeast and Lost Lake on the island in the center of the San Jacinto River west of Lynchburg Reservoir (Figure 3-1). Upland, riparian, and aquatic habitats are present.

3.4.1 Upland Habitats

Upland natural habitat adjacent to the San Jacinto River in the Site vicinity is generally low-lying, with little topographic variation, and consisting primarily of clay and sand that supports loblolly pine-sweetgum, loblolly pine-shortleaf pine, water oak-elm, pecan-elm, and willow oak-blackgum forest communities along the river's banks (TSHA 2009). Upland natural habitat occurs along narrow sections of land on either side of the river, as well as on several small islands, to the north and south of I-10 and east of the impoundments. Most of these islands are vegetated with a mixture of shrubs and trees, with fringing shallow waters. These habitats could support mammals, such as marsh rice rats and deer, that could migrate to the islands close to mainland areas, as well as passerines that could use the vegetated uplands for nesting and foraging, and shoreline birds such as sandpipers and herons that could wade and forage in the shallow areas adjacent to the islands.

Uplands on the western edge of the site north of I-10 are generally less densely developed than across the river along the Site's eastern border, which is developed with a mix of residential and commercial land uses (Figure 3-2). The I-10 freeway fragments the natural areas to the north and south of the highway, reducing the connectivity of these habitats. On the peninsula to the south of I-10, most of the upland habitat is zoned for commercial or industrial use, with the exception of a narrow segment of land on the western edge of the Site south of I-10 (Figure 3-2). The upland vegetation present on the southern peninsula is primarily low-lying grasses, with a few shrubs and trees adjacent to the shoreline.

3.4.2 Upland Wildlife

There is no site-specific data describing wildlife uses of the upland portions of the Site. Based on local wildlife lists and the types of habitat and land uses present at the Site, it is reasonable to expect a suite of generalist terrestrial species that are not highly specialized in their habitat requirements and are adapted to moderate levels of disturbance. The reptiles and amphibians that could occur in the vicinity of the Site include snakes, alligators, and turtles (Table 3-3). Avian taxa using upland habitats may include sparrows and other generalist passerines, starlings, pigeons and doves, corvids, and killdeer. Mammals expected in a semi-urban environment like the Site include small mammals (rodents), skunks, raccoons, coyotes, and opossums.

3.4.3 Aquatic and Riparian Habitats

Habitats on the northern portion of the Site include shallow and deep estuarine waters, and shoreline areas occupied by estuarine riparian vegetation. Because the Site is within an estuary, the salinity of the San Jacinto River in the vicinity of the Site can be low at times

(1 to 5 parts per thousand [ppt]; Clark et al. 1999); it was 2 to 12 ppt in a recent study (University of Houston and Parsons 2009). The in-water portion of the Site is unvegetated, with a deep (20- to 30-foot) central channel, and shallow (3 feet or less) sides (NOAA 1995; Clark et al. 1999). Except in the impoundments north of I-10, sediments are sandy and characterized by low organic matter content; most surface sediment samples collected within the northern impoundment ranged between 1 and 5 percent total organic carbon (TOC), and TOC in samples collected from within USEPA's preliminary Site perimeter but outside the impoundments was lower, with most samples between 0.5 and 2 percent TOC (Integral and Anchor QEA 2012) and having a high sand content. In surface sediment samples collected on the Site, the fraction consisting of sand ranged from 4 to 98 percent, with an average of about 50 percent sands.

3.4.3.1 Fish and Invertebrates

The tidal portions of the San Jacinto River and upper Galveston Bay provide rearing, spawning, and adult habitat for a variety of marine and estuarine fish (Table 3-4) and invertebrate species (Table 3-5). Species known to occur in the vicinity of the Site include clams and oysters, blue crab (*Callinectes sapidus*), black drum (*Pagonius cromis*), southern flounder (*Paralichthys lethostigma*), hardhead (*Ariopsis afelis*) and blue catfish (*Ictalurus furcatus*), spotted sea trout (*Cynoscion nebulosis*), and grass shrimp (*Paleomonetes pugio*) (Gardiner et al. 2008; Usenko et al. 2009).

3.4.3.2 Shorelines and Wetlands

A sandy intertidal zone is present along the shoreline throughout much of the Site (Figure 2–1). Minimal habitat is present in the upland sand separation area, as demolition and closure of this area created a denuded upland with a covering of crushed cement and sand. The sandy shoreline of the sand separation area is littered with rip-rap, metal debris, and piles of cement fragments. An estimated 34 acres of estuarine and marine wetlands are found within USEPA's preliminary Site perimeter (Figure 3-3). Throughout the broader surrounding area, there are approximately 55 additional acres of freshwater, estuarine, and marine wetlands (Figure 3-1).

A wetland delineation for areas of the Site to the north of I-10 completed in 2010 prior to implementation of the TCRA (BESI 2010) identified a large portion of the area within the 1966 northern impoundment perimeter above high water as emergent intertidal wetlands. In addition, some patchy areas with wetland characteristics were identified around the margin of the northern impoundments, most of which were narrow in width and a few hundred feet in length, including fringing wetlands between the open water of the San Jacinto River and upland portions of the Site, and emergent wetlands associated with roadside ditches north of I-10 (Figure 3-3). Major vegetation found in association with fringing wetland areas included broadleaf cattail (*Typha latifolia*), saltmeadow cordgrass (*Spartina patens*), saltmarsh aster (*Symphyotrichum divaricatus*), marshelder (*Iva annua*), and saltgrass (*Distichlis spicata*). Other aquatic and wetland plants that could occur in the wetland habitats on the Site are listed in Table 3-6. The vegetation associated with the estuarine intertidal wetland documented on the north impoundment (Figure 3-3) is no longer present on the site as a result of the TCRA (Figure 2-3), discussed further below.

Wetland habitats to the south of I-10 along the eastern side of the channel include a narrow stretch of vegetation along the shoreline and the shoreline habitats of three small islands south of I-10. The vegetation on the islands mainly consists of shrubs and small trees. The shrubs and small trees which overhang the water line may provide some shelter and in-water habitat structure for juvenile and baitfish. This area also provides limited foraging habitat for mammals such as raccoons, opossums, skunks, and birds.

3.4.3.3 Aquatic Wildlife

Aquatic birds and semiaquatic mammals that are found in the vicinity of the Site include ducks, shorebirds, wading birds (herons and egrets), diving piscivores, and various others (Table 3-7). There are a number of migratory bird species known to winter in the vicinity of the Site. They include belted kingfisher (*Megaceryle alcyon*), red breasted merganser (*Mergus serrator)*, greater yellowlegs (*Tringa melanoleuca)*, western sandpiper (*Calidris mauri*), and dabbling ducks including gadwall (*Anas strepera*) and teal. Herons and closely related birds that use wetland and estuarine habitats and that may be present in the Site vicinity include the green (*Butorides virescens*), tri-colored (*Egretta tricolor*), and little blue (*E. cerulea*) herons, and also the black-crowned (*Nycticorax nycticorax*) and yellow-crowned

(*N. violacea*) night-herons. Raptors, rails, pelicans, gulls, ducks, and sandpipers are also among the aquatic-dependent and aquatic-associated bird species that use the aquatic habitat that is present in the vicinity of the Site. Sandpipers, egrets, and herons are wading birds that forage along shallow intertidal areas for benthic macroinvertebrates and small fish. Piscivorous bird species that may forage in the open waters of the river include cormorants, osprey, and pelicans. Omnivores including gulls and ducks may forage at the river's edge as well as in the water column. Mammals using both aquatic and wetland habitats that could occur in the vicinity of the Site include the marsh rice rat, muskrats, nutria, and raccoon (Table 3-8).

3.4.3.4 Effect of TCRA Construction

Prior to implementation of the TCRA, estuarine riparian vegetation lined the upland area that runs parallel to I-10 to the north. As a result of the TCRA, that area now includes a dirt road. The western cell of the waste impoundments north of I-10 was occupied by estuarine riparian vegetation until the recent implementation of the TCRA, when the vegetation was removed (Figure 2-3). The eastern cell, also completely covered as a result of the TCRA, lies within intertidal and subtidal habitats.

Under baseline conditions (prior to the implementation of the TCRA), the estuarine riparian vegetation present on the western cell was made up of a mixed shrub and tree canopy (Figure 2-3), could have provided habitat for foraging, nesting and shelter to a variety of bird and mammal species (Tables 3-7 and 3-8). Prior to TCRA implementation, clam shells were observed on the site, indicating a food source for animals such as raccoons, coyotes, wading birds, gulls, and corvids. As part of the TCRA construction, nearly all vegetation was removed from the entire western cell (Figure 2-3), leaving only small amounts of plant material on the western edge of the cell, and eliminating opportunities for upland foraging, nesting and refuge. The shoreline habitat in the TCRA footprint is now devoid of cover, and the exposed surface has limited habitat value for birds and small mammals, and in turn by their predators, such as coyote and raccoon. Some shoreline wading birds may still be expected to use the sandy, shallow intertidal zone for foraging in the post-TCRA conditions. However, over time the area affected by the TCRA cap would be expected to undergo some sedimentation, resulting in the development of plant habitat and plant community development, and subsequent use of the area by birds and mammals. Wildlife uses of this area in the future could be very similar to those of the baseline condition.

3.4.4 Endangered and Threatened Species at the Site

Wildlife that are state-listed as threatened and endangered and have the potential to be found in the general vicinity of the Site are:

- Timber rattlesnake
- Smooth green snake
- Alligator snapping turtle
- White-faced ibis
- Brown pelican
- Rafinesque's big-eared bat.

In addition to these listed species, the American bald eagle, protected under the federal Bald and Golden Eagle Protection Act and listed as threatened by the State of Texas, may be found in the vicinity of the Site.

The two snakes that are listed above are unlikely to occur on the Site. Available information on habitat for these snakes indicates that they prefer upland forested habitats, prairies, and fields or mesic habitats with good vegetative cover. They are not considered common occupants of estuarine or marine wetlands.

The alligator snapping turtle is found in a variety of aquatic habitats including lakes, oxbows, deep rivers, creeks, ponds and brackish estuaries (Appendix A). This species is an opportunistic carnivore, feeding primarily on fish but also on a range of other aquatic animals and occasionally aquatic plants. They spend most of their time in water, usually in the deepest part of their habitat. They are primarily a freshwater species, though they may occasionally use low salinity environments (Appendix A). It is therefore possible that alligator snapping turtles may use aquatic habitats in the Site vicinity, even though their use of the Site is expected to be low relative to their use of freshwater habitats.

The white faced ibis prefers freshwater wetlands, but can be found in estuarine habitats. It is intermediate to the surrogate receptors sandpiper and great blue heron in terms of both body size and diet. Its foraging strategy of visual hunting and tactile probing of sediments for invertebrates is similar to that of the sandpiper, making sandpiper an appropriate representative for this species (Ryder and Manry 1994). The ibis is omnivorous and opportunistic, consuming aquatic insects (in freshwater), fish, amphibians, and crustaceans. The extent to which this bird would use the Site is unknown, but it has been observed as on occasional visitor in summer and fall and rarely in winter and spring at the nearby Baytown nature center. There is limited information regarding the home range or foraging range of this species, though an estimate based on expert opinion and limited information about dispersal to foraging sites suggests that this species may require habitat patches greater than 12 km^2 (Appendix A). The type of habitat present at the Site is not like the foraging and nesting habitats preferred by the ibis, which primarily forages in shallow freshwater habitats with emergent vegetation like rushes and cattails (Ryder and Manry 1994). The white-faced ibis would be only an occasional visitor to the Site, and its exposure potential is considered low. Because sandpiper is assumed to use only the Site while the ibis is an occasional visitor, the sandpiper is a conservative representation of shorebirds such as the ibis.

The brown pelican is a marine piscivore that preys on small surface-schooling fishes. The brown pelican may range up to 20 km from nesting colonies during the breeding season and as far as 75 km from the nearest land during the non-breeding season (Shields 2012). Its diet is similar to that of the neotropic cormorant, making neotropic cormorant an appropriate representative for this species. Although there is little information regarding the foraging area of brown pelican, and information was insufficient to estimate a home range, given the wide-ranging, pelagic nature of the pelican, it is reasonable to assume that its foraging area is likely to be greater than the area of the Site used by the neotropic cormorant.

The American bald eagle may hunt for fish, or eat carrion found on terrestrial and shoreline areas. Foraging ranges for the bald eagle vary widely, from less than 10 km^2 to thousands of square kilometers depending on season and breeding status of the bird (Appendix A). The great blue heron is an appropriate representative for the bald eagle, as it is an omnivore feeding on a range of fish prey. In addition, the great blue heron's foraging strategy and diet make it likely that its association with sediments and rate of incidental sediment ingestion is likely to be higher than that of the bald eagle, making it a conservative choice for evaluating risks to bald eagle from the sediment exposure pathway.

The Rafinesque's big-eared bat may possibly use bridge structures or abandoned buildings in the vicinity of the Site for roosting (TPWD 2012c), but is not expected to forage in the habitats found in the vicinity of the Site because it feeds primarily on emergent aquatic insects, which are generally restricted to freshwater systems and are uncommon in brackish estuarine waters.

In light of this information the white-faced ibis, brown pelican, and American bald eagle are the protected species with a reasonable likelihood of occurring and possibly foraging on the Site.. Because the white-faced ibis, pelican, and eagle have foraging ranges significantly greater than the range of the selected surrogates, and greater than the area of the Site, the selected surrogates represent a much greater exposure potential than these three species. Therefore, risk to these species is considered negligible for a given $COPC_E$ when all of the avian receptor surrogates have negligible risks for that $COPC_E$. In cases where risk could be present for the surrogate species, differences between the home range of each of these protected species and the exposure unit used for modeling exposure to the surrogates provide the basis for evaluation of exposure and risk to the protected species. The method for the exposure evaluation is described in Section 4.3.1.6, and the approach to interpretation of results in presented in Section 6.1.

3.5 Ecological Receptors and Receptor Surrogates

Selection of receptors for this BERA was documented in the SLERA (Appendix B to the RI/FS Work Plan), and summarized in this section. Ecological receptors for the south impoundment are selected in Appendix E to this document.

Ecological receptor surrogates are selected to be representative of the trophic and ecological relationships known or expected at the Site. In selecting receptor surrogates for evaluation in the BERA for the Site, the following criteria were considered:

- The receptor is or could potentially be present at the Site.
- The receptor is representative of one or more feeding guilds.

- The receptor is known to be either sensitive or potentially highly exposed to COPCEs at the Site.
- Life history information is available in the literature or is available for a similar species that can be used to inform life history parameters for the receptor.

Many species of aquatic-dependent wildlife may nest in, forage in, and/or migrate through the lower San Jacinto River system. Detailed listings of the species of plants, benthic invertebrates, reptiles, fish, birds, and mammals that could use the habitats on the Site or in the vicinity of the Site are provided in Tables 3-3 through 3-8.

Given that sediments, upland soils, and surface water are the primary environmental media determining the fate and transport of Site-related chemicals, the choice of receptors focused on aquatic-dependent species, or those species which use aquatic resources to a substantial extent. Fish and aquatic-dependent wildlife species for which there are potentially complete exposure pathways to Site-related chemicals include those with direct contact with contaminated soil, sediment, and water and those that prey on benthic macroinvertebrates or on fish that consume benthic macroinvertebrates. Few amphibians that are potentially present in the region are tolerant of brackish or saline waters, with the possible exception of the southern leopard frog. Amphibians are therefore not likely to be in contact with contaminants at the Site, are probably not an ecologically important component of the ecosystem expected at the Site, and are not considered relevant to the BERA.

Terrestrial species are also represented by avian and mammalian surrogate receptors that use upland habitats. The receptors selected for this BERA to address ecological risks for the north impoundment and surrounding aquatic environment are summarized in Table 3-9. More detailed discussion of their life histories in support of evaluating exposures is provided in Appendix A.

3.6 Assessment Endpoints and Risk Questions

An assessment endpoint is "an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes" (USEPA 2003b). An assessment endpoint addresses a value of ecological significance, has an unambiguous operational definition, and is readily measured or predicted (Suter 1993). Ecological properties identified in assessment endpoints should be those which are susceptible to the chemical stressors and relevant to management goals. Clearly defined assessment endpoints help structure the assessment to address risk management and the primary concerns of stakeholders. Assessment endpoints discussed in this section were derived to conform to these guidelines. A summary of assessment endpoints and risk questions for each receptor group addressed by this SLERA is provided in Table 3-10.

The available literature, and the specific types of information it provides, determine to some extent how the assessment endpoints are defined. For example, although a wildlife *population* is often the level of ecological organization of significance to management, literature to provide measures of effects generally reports on effects on *individuals*. Metrics to assess attributes of individuals (i.e., individual survival, growth, and reproduction) are more generally available in the literature used to support ecological risk assessment than metric to address populations. As a result, some of the assessment endpoints presented in the SLERA have been slightly modified to more closely reflect the ecological attributes that are more commonly reported in the available toxicity literature for the COPCES, and to link the attributes addressed (i.e., individual-level) to the attributes relevant to risk management (i.e., population- or community-level). The fundamental ecological values expressed by the assessment endpoints have not been changed.

In the absence of site-specific population data, performing a series of actual population assessments for an ERA at a site is generally impractical. Because assessment of exposure and effects in this BERA relies on models, the assessment endpoints, for receptors other than the protected species, generally are population viability as indicated by survival, growth, and reproduction of individuals. Units for exposure estimates are selected to match the expression of exposure used in the available toxicity literature.

3.7 Ecological Conceptual Site Models

In the context provided by the CSMs, the receptors and exposure routes evaluated by this BERA include:

- The benthic macroinvertebrate community exposed through direct contact with the benthic environment (sediment, porewater, and surface water)
- Bivalve molluscs exposed through direct contact with the benthic environment (sediment, porewater, and surface water)
- Fish (in all feeding guilds) exposed through ingestion of sediment and food, and respiration of water
- Reptiles exposed through ingestion of sediment or soils, water, and food
- Birds (in all feeding guilds) exposed through ingestion of sediment or soils, water (for seabirds only), and food
- Mammals exposed through ingestion of sediment or soils and food.

Table 3-11 outlines each line of evidence used to address risk to these taxa and exposure groups.

3.8 Ecological Risk Analysis Plan

The problem formulation provides a complete description of the context for evaluation of ecological risks at the Site. In this context, this BERA uses standard methods provided for by USEPA guidance (USEPA 1997, 1998, 2008), including evaluation of uncertainty for results that reflect both reasonably conservative assumptions and realism when Site-specific information is available. This section provides a synopsis of the risk assessment approach and methods used to address the assessment endpoints and risk questions listed in Table 3-10. The purpose of this section is to summarize the overall approach to the risk assessment, to identify the measures of exposure and effects used, to outline the analytical steps for each selected analysis tool, to describe the approach to compilation of information on potential effects to ecological receptors, and to identify the means to characterize risk and evaluate uncertainty. Subsequent sections report the specific calculations, assumptions and related selection of data for the computation of exposure parameter estimates and supporting rationale, and report the outcome of each analysis step.

3.8.1 Ecological Risk Assessment Approach

According to USEPA guidance, a baseline risk assessment should be realistic, so that results accurately represent risks at the Site prior to remedial action (USEPA 1988). Unlike a screening level evaluation, the conclusions of a BERA should reflect realistic representations of exposure and toxicity, and the supporting analysis should not be overly conservative. This approach is appropriate because the baseline risk assessment results are used to inform selection of a risk management approach that is cost-effective.

This BERA uses a tiered approach to the risk analysis and characterization of risks under baseline conditions: an initial assessment of risk is performed using a deterministic model for each receptor and each $COPC_E$ to which that receptor may be significantly exposed via a major exposure pathway ("receptor–$COPC_E$ pair"). The initial assessment employs reasonably conservative but realistic assumptions for each receptor and exposure pathway. In some cases, screening level benchmarks are used for comparison with Site-specific data. This reasonable worst case analysis provides a gross evaluation of risk, resulting in a hazard quotient (HQ) for each receptor–$COPC_E$ pair. HQs are calculated for each receptor–$COPC_E$ pair using a no-observed-adverse-effects level (NOAEL) for the $COPC_E$ to derive the HQ_N, and a LOAEL for the HQ_L. HQs are reported to one significant figure. For each receptor–$COPC_E$ pair, the need for risk analyses subsequent to calculation of the HQ depends on the value of the HQ_L, with one of three possible outcomes, as shown in Figure 3-4. Interpretation of HQs is described in Section 3.8.4.1, below.

When the HQ_L is equal to or greater than 1, subsequent analyses include:

- A probabilistic exposure evaluation
- Evaluation of post-TCRA risk
- Consideration of background.

These three analyses are performed to support risk management decision-making. Use of probabilistic assessment tools results in a more complete and transparent characterization of risks and uncertainties than is possible using an HQ alone. The post-TCRA risk condition existing on the Site is evaluated for those receptors considered to have an unacceptable risk under baseline conditions. Although the TCRA is not considered part of the baseline condition, the purpose of this evaluation is to see what impact the TCRA has on ecological risk, using the general assumption that $COPC_E$ concentrations in sediments within the TCRA footprint are equal to the median concentration of the chemical in the upstream background sediment dataset. This information will inform consideration of the TCRA in the evaluation of remedial alternatives in the FS. Background ecological risks are characterized to describe the incremental risks due to the Site.

An overview of each step in the analytical approach is provided below, with additional details provided in subsequent sections.

3.8.2 Exposure Assessment

According to the CSM, aquatic receptors may be exposed to COPCES via transport of dissolved chemicals across the gills, ingestion, and direct contact. In many cases, the specific route of exposure cannot be discerned from the available literature, or it is not important to the interpretation of the potential for toxicity, because exposures in the literature are expressed simply as concentrations in water, sediment, or organism tissue (see Appendix B of the RI/FS Work Plan, Section 4). Exposures to birds, mammals, and reptiles occurring through respiration (inhalation) or dermal absorption are not evaluated in the BERA as these are generally considered to be minor pathways of exposure relative to the ingestion pathway (USEPA 2003a), although there is uncertainty about this assumption for reptiles (Weir et al. 2010).

3.8.2.1 Measures of Exposure

Measures of exposure selected to address benthic macroinvertebrate and fish receptors include concentrations of COPCs in the following general categories:

- Surface water (mg/L)
- Bulk sediment (mg/kg dry weight [dw])
- Tissue of whole fish, or benthic macroinvertebrates (mg/kg wet weight [ww]; mg/kg lipid weight).

Measures of exposure selected to address bird, reptile, and mammal receptors were the concentrations of COPCs in the following general categories:

- Surface water (mg/L)
- Sediment (mg/kg dw)
- Soils (mg/kg dw), for terrestrial receptors

- Tissue of whole fish (mg/kg dw)
- Tissue of benthic macroinvertebrates (mg/kg dw)
- Bird egg tissue (mg/kg ww; mg/kg lipid weight), estimated from concentrations in diet of birds.

To the maximum extent possible, empirical Site-specific data are used to compute the EPCs for each of these measures of exposure. In some cases, modeling is required to derive exposure concentrations. Models are used to estimate $COPC_E$ concentrations for the following:

- Surface water concentrations of $COPC_E$s other than dioxins and furans (because empirical data are available for dioxins and furans)
- Terrestrial invertebrate prey and plant foods ingested by killdeer, marsh rice rat, alligator snapping turtle, and raccoon
- Concentrations of dioxin, furan, and dioxin-like PCB congeners in bird eggs.

The specific models and datasets used to derive estimates for these parameters are provided in Section 4.

3.8.2.2 Exposure Units and Calculation of Exposure Point Concentrations

As for the human health exposure assessment (Integral 2012), exposure units are identified for each ecological receptor prior to calculation of EPCs. An exposure unit reflects the area within which a receptor may contact an exposure medium. Spatially defined exposure areas are used to identify the specific set of samples needed to calculate the EPCs for each exposure unit.

For each $COPC_E$ in each exposure medium within each exposure unit, an expression of the central tendency (CT) of the dataset, and an expression of the reasonable maximum (RM) concentration are prepared. The means to estimate these two expressions of concentration depend on the distribution of the dataset, and may include the mean, median, or other expression for the CT; and the 95 percent UCL (95UCL), 95th percentile, or maximum for the RM. Using these two expressions of the EPC for any given $COPC_E$ enables presentation of the most likely (CT) exposure, along with the upper bound (RM) exposure condition, and reflects the middle and upper extent of the exposure profile for receptors. This profile is biased high, because the reasonable minimum (RMin) calculated as the 95 percent lower confidence limit on the mean (or a similar statistic) is just as likely to occur as the RM. An illustration of the importance of this bias in interpreting risks calculated with the RM is provided in Section 7.

For the probabilistic exposure assessment, the probability distribution of the chemical or TEQ concentration is derived from Site-specific data. For parameters estimated probabilistically but with no Site specific information, simple assumptions were made about the data distributions using information from the literature. Details are provided in Section 4.

3.8.2.3 Exposure Algorithms

Following derivation of EPCs for each exposure medium, one or more of the following are performed for the deterministic risk evaluation:

- Concentrations in water, sediment, or tissue are directly compared to benchmarks and/or TRVs expressed in the same exposure units.
- Concentrations in various exposure media are integrated for an individual receptor to compute a cumulative (for all exposure media) total daily ingestion rate.

For the latter, standard exposure algorithms commonly used in ecological risk assessments are used, and described in detail in Section 4. If a probabilistic evaluation of exposure is required, Site-specific data are used to define the probability distribution of several exposure parameters, including life history parameters and the concentrations of the $COPC_E$ in each exposure medium within each exposure unit.

3.8.2.4 Exposure Assumptions

Exposure algorithms for birds and mammals require a number of assumptions about the aspects of receptor biology that affect $COPC_E$ exposure, such as body weight, food or soil ingestion rate, and home range area. Estimates of relevant exposure parameters were taken from the primary literature or from USEPA's (1993) Exposure Factors Handbook. Information on the habits and life history of each receptor is provided in Appendix A, and a summary of related exposure parameters used in exposure algorithms is provided in Table 3-12.

3.8.3 Effects Assessment

This BERA relies on published literature to identify and describe toxicity of COPCES to ecological receptors. No Site-specific toxicity studies were conducted in support of this BERA. The effects assessment therefore consists of a review and compilation of TRVs and benchmarks. For the purposes of this document, a TRV is a species-specific value derived from a controlled experimental study at environmentally realistic exposure levels. The study underlying the TRV defines the specific type of toxic effect at a defined exposure level and exposure route or condition. A TRV can be either an NOAEL (or analogous, e.g., no-observed-effects concentration) at which no effect is expected, or an LOAEL (or analogous), which is the threshold level at or above which effects are expected. A benchmark is a derived value that reflects a broad array of information, potentially encompassing several species, and considered generally protective of a group of species or a community type. An example of a benchmark is USEPA's ambient water quality criteria for the protection of aquatic life (AWQC). Either a TRV or a benchmark can be considered a measure of effect.

The individual effects measures derived from the literature were those that could clearly be related to population- or community-level effects, consistent with the selected assessment endpoints (Table 3-10). Each selected measure of effects on ecological receptors addresses changes in survival, growth, or reproduction resulting from exposure to one or more COPCES. Survival, growth, and reproduction (including developmental inhibition leading to juvenile mortality) can clearly be related to population impacts. For invertebrates, the literature and some benchmarks address higher levels of organization such as populations and communities. Studies addressing endpoints below the organism level (e.g., cellular or biochemical alterations or gene expression), which are difficult or impossible to relate to population- or higher-level effects, were not used to establish TRVs for the BERA.

When using published toxicity literature to establish measures of effect, the specific effects measure depends on the experimental design that was used. For example, a toxicity study may provide a threshold dose above which a reduction in the hatchability of bird eggs occurs. In this case, the effect category is reproduction, and the measure is the LOAEL at or above which effects are observed. TRVs, which encompass both LOAEL and NOAEL values, can be expressed in several ways. The methods for selecting TRVs and benchmarks as well as the values used in this BERA are summarized in Section 5 and described in detail in Appendix B.

3.8.4 Risk Characterization and Uncertainty Analysis

One or more measures of exposure and one or more measures of effects are used to address each assessment endpoint and address the risk questions related to that endpoint. Measures of exposure and effect together define each line of evidence to address each assessment endpoint. Table 3-11 provides a summary of each receptor, assessment endpoint and the measures of exposure and effects that will be used for each line of evidence.

3.8.4.1 Calculation of Hazard Quotients

For each receptor–$COPC_E$ pair, risks are initially described using an HQ, calculated as the estimated exposure as an environmental concentration or daily dose based on the CT of the exposure data distribution, divided by the measure of effect. The ratio of the exposure estimate to the TRV or benchmark is calculated using the following equation:

$$HQ = E \div TRV \qquad \text{(Eq. 3-1)}$$

Where:

HQ	=	hazard quotient
E	=	estimated exposure
TRV	=	toxicity reference value or benchmark.

Units used for the exposure estimate and for the TRV may vary among lines of evidence, but must be the same for the numerator and denominator in the HQ equation. Individual HQs are calculated for each chemical, or TEQ.

To interpret results of HQ calculations, the following guidelines are used:

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level lower than the NOAEL (i.e., $HQ_N < 1$) is characterized as negligible.

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level between the NOAEL and LOAEL (i.e., $HQ_N > 1 > HQ_L$) is characterized as very low, and is discussed in the context of the toxicity data supporting the NOAEL and LOAEL values.
- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level higher than the LOAEL (i.e., $HQ_L > 1$) is considered to be present. Risk to the assessment endpoint, which may be a population or community, is evaluated and discussed further in the context of the data supporting the TRV.

Measures of effect (TRVs) typically describe effects on individuals. To inform the risk assessment, HQs must be interpreted to describe risk to the assessment endpoint, which is generally a community or population of organisms (Table 3-10). Therefore, interpretation of $HQ_N > 1$ when $HQ_L < 1$ involves professional judgment and is informed by the basis for the TRV used in the HQ calculation. In these cases, conclusions about risk incorporate relevant context about assumptions and the toxicity information supporting the calculation. Supporting information is described in risk conclusions in Section 6.

Additivity of toxicity and risk for an individual receptor exposed to multiple chemicals (other than dioxin-like compounds) is not systematically considered or reported in this BERA. The absence of relevant information to address this issue is discussed in Section 7.

3.8.4.2 Probabilistic Risk Evaluation

A probabilistic evaluation of exposures is performed for ecological receptors that are potentially exposed at levels equal to or greater than the effects threshold according to one or more lines of evidence in the deterministic evaluation. The probabilistic assessment assigns probability distributions for exposure parameters to yield an output probability distribution for the exposure estimate. Risks to receptors with $HQ_L > 1$ are characterized as the probability that an individual (conforming to the exposure scenario represented by the exposure assumptions) is exposed at or above a level known to have a specified effect. This method allows risk to be expressed as the likelihood that exposures associated with adverse effects can occur under the exposure assumptions. An example of a risk statement of this type is: "there is a 3 percent probability that raccoons will be exposed to iron at a level that has been observed to result in the reduced growth of juvenile mammals."

3.8.4.3 Evaluation of Post-TCRA Risk

For ecological receptor–$COPC_E$ pairs for which $HQ_L > 1$, the post-TCRA risks are also considered. To evaluate the degree to which implementation of the TCRA reduced the baseline ecological risks, values for exposure parameters associated with sediment and soil that were capped by the TCRA are recalculated using a simple framework. For calculation of post-TCRA EPCs, sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of I-10 are eliminated from the dataset used to estimate EPCs, and replaced with the median concentration of the chemical in the upstream background sediment dataset or from the background soil dataset, as appropriate. Samples collected outside the 1966 impoundment perimeter are considered to be the same as for the baseline condition. This approach assumes that birds and mammals that could be using the area affected by the TCRA cap under baseline condition will use the area the same way in the future, and that concentrations of COPCs in sediment in the future will be equivalent to the background condition established for the RI.

3.8.4.4 Risks to Populations of Ecological Receptors

Population-level and community-level assessment endpoints have been selected for the BERA, consistent with USEPA guidance (USEPA 2003b), but TRVs from the available literature providing measures of effects generally represent individual-level endpoints (i.e., those related to survival, growth and reproduction of individual organisms), particularly for birds and mammals. Population-level assessment endpoints are addressed qualitatively in the risk conclusions.

3.8.4.5 Comparison of Site Risks to Background

Background ecological risks are characterized using data from background areas to provide perspective on risks associated with the Site, and to gain an assessment of the incremental risks due to the Site. Only the incremental increase in risk relative to background can potentially be directly affected by controls at the Site.

Background risks are not calculated for all receptors and $COPC_{E}s$, but are performed when it is concluded that there is an unacceptable baseline risk to an assessment endpoint from a $COPC_{E}$. Evaluation of risks in background areas was conducted using the same general lines of evidence as for evaluation of Site specific risks. The Site-specific background dataset generated for this RI is used. A summary of that dataset is listed in Section 2.2.1. Details describing background sampling are provided in the Field Sampling Reports (FSRs) for this project, submitted to USEPA in July 2011 (Integral 2011e; Integral and Anchor QEA 2011a,b). Additional background sediments were collected in 2011 and are not described in the FSRs; a complete description of the RI data set will be provided in the RI Report.

4 EXPOSURE ASSESSMENT

Measures of exposure include concentrations of individual COPCEs in water, foods of fish and wildlife, sediment, soil and the eggs of birds, and daily ingestion rates of COPCEs for reptiles, birds and mammals. Because some of the fundamental concepts and resulting selection of methods differ substantially by receptor group, this section is organized by receptor (or receptor group). Each subsection presents methods including algorithms and supporting assumptions, provides a summary statement of the data used, and presents results of the exposure assessment to address each line of evidence for the receptor. Summary information to describe the results of the exposure assessment is presented for each receptor group at the end of each subsection, or in an appendix.

4.1 Exposure of Benthic Macroinvertebrates

Lines of evidence to evaluate risk to benthic macroinvertebrates include:

1. Comparison of bulk sediment concentrations of each COPCE to literature-based benchmarks or TRVs expressed as a concentration in sediment
2. Comparison of estimated concentrations of each COPCE in porewater to literature-based benchmarks or TRVs expressed as a concentration in water
3. For dioxins and furans only:
    a. Comparison of the concentration of 2,3,7,8-TCDD in tissue of whole clams to critical tissues residues (CTRs) expressed as a concentration in tissue of molluscs
    b. Comparison of the concentration of 2,3,7,8-TCDD in sediments with an NOAEL for sediment.

For all of the COPCEs, the primary line of evidence is comparison of sediment concentrations at individual stations to benchmarks or TRVs expressed as a bulk sediment concentration. For any COPCE for which the first line of evidence could not be used because a benchmark or TRV is not available in that form, a TRV expressed as a concentration in water is used, and compared to estimated porewater concentrations for that chemical. Comparison of the concentration of 2,3,7,8-TCDD in tissue collected from the Site with the CTR for molluscs is only used to address risks of dioxin exposure to clams. For these lines of evidence, exposure to benthic macroinvertebrates can be characterized as a concentration in sediment, porewater, or tissue, and each approach has a different spatial context:

- Lines of evidence 1 and 3b are empirically based, and describe exposure to benthic macroinvertebrates at each sampling station where a surface sediment sample was collected.
- Line of evidence 2, which is only used for those chemicals lacking benchmarks or TRVs expressed as bulk sediment concentrations, requires an estimate of porewater concentration for each $COPC_E$. To evaluate risk to benthic invertebrates on the basis of exposures via sediment porewater, the water concentration is estimated using the concentration of each $COPC_E$ in sediment at each sampling location.[3]
- Line of evidence 3a addresses molluscs only, and relies on empirical data for 2,3,7,8-TCDD in clam tissue, representing small groups of organisms (those making up a composite). For clams, the composite of several individuals into a single tissue sample represents exposure across a small area.

None of these lines of evidence are reasonable for two metals, aluminum and vanadium, because their geochemistries in estuarine environments strictly limit their bioavailability and toxicity to benthic invertebrates (discussed further below). Concentrations of each of these two metals at each surface sediment sampling station were compared to their respective reference envelope values (REVs).

4.1.1 Estimated Porewater Concentrations

To evaluate exposure of benthic macroinvertebrates to $COPC_E$s via porewater, methods based on principles of equilibrium partitioning are used to extrapolate bulk sediment concentrations of each $COPC_E$ to estimate a water concentration. By using very conservative parameters, this method provides an upper limit estimate of porewater concentrations. It assumes that the sediment porewater is a limited volume of water in direct contact with sediment solids, and is in a two-phase equilibrium with the sediment solids.

[3] Empirical data are available to describe concentrations in water for dioxins and furans, but these data were not used because the first and third lines of evidence are preferred for assessment of risk to benthic macroinvertebrate communities due to TCDD.

The methods to extrapolate from sediment to porewater are as follows. For the metals that are $COPC_E$s for benthic invertebrate communities but that lack TRVs expressed as bulk sediment concentrations, the concentration of each $COPC_E$ in porewater was estimated from the sediment concentration and its respective soil–water partitioning coefficient, or K_d (Table 4-1):

$$C_{PW} = C_s \div K_d \qquad \text{(Eq. 4-1)}$$

Where:

C_{PW}	=	concentration in porewater (mg/L)
C_S	=	concentration in sediment (mg/kg dry weight)
K_d	=	sediment–water partitioning coefficient (L/kg dry weight).

For organic $COPC_E$s, the role of organic carbon is considered. Because organic carbon binds to many organic chemicals, it can alter the rate of partitioning from the sediment matrix into water and limit the amount of the COPC that can be dissolved into water or porewater. For organic $COPC_E$s, the following algorithm was used to first convert the dry weight sediment concentration to an organic-carbon normalized concentration for each sample:

$$C_{s,oc} = C_s \div f_{oc} \qquad \text{(Eq. 4-2)}$$

Where:

$C_{s,oc}$	=	organic carbon (OC)-normalized concentration in the sediment (mg/kg dw)
C_s	=	sediment concentration of the specific COPC (mg/kg dw)
f_{oc}	=	fraction of organic carbon in the sediment sample (unitless).

The organic carbon-normalized concentration for each sample is then used to estimate porewater concentrations at that sample location as follows:

$$C_{pw} = C_{s,oc} \div K_{oc} \qquad \text{(Eq. 4-3)}$$

Where:

C_{pw}	=	concentration in water (mg/L)
$C_{s,oc}$	=	organic carbon-normalized concentration in sediment (mg/kg OC)
K_{oc}	=	organic carbon–water partitioning coefficient (L/kg OC).

Values for K_{OC} are provided in Table 4-1. To evaluate risk to macroinvertebrates that make up benthic infaunal communities, porewater concentrations are estimated for each sediment station from the individual $COPC_E$ concentration in sediment at that station. Porewater concentrations of organic $COPC_Es$ for each station were estimated using the fraction of organic carbon in sediment at that station to express sediment concentrations on an organic carbon basis, per Equation 4-2.

Overall, the use of sediment concentrations and partitioning coefficients derived for soils (used for metals) to estimate porewater concentrations is highly conservative. For two of the metals, aluminum and vanadium, the use of partitioning coefficients derived for soils (RAIS 2011) to estimate estuarine porewater concentrations was considered inappropriate because of the geochemical conditions in the estuarine environment. While most COPCs are trace constituents, aluminum is a major rock forming mineral that is an important constituent of clay and feldspar minerals that constitute a large fraction of the inorganic constituents of the sediments. As a reference, aluminum is the third most abundant mineral in both the Earth's crust in general (Krauskopf 1979) and sediments in the nearby Mississippi Delta (Clark 1924) following only oxygen and silica. As a rock forming mineral, aluminum concentration is controlled by mineral solubility, which is affected by the sediment composition and pH. Generally, at neutral pH, aluminum solubility is very low (less than 1 µg/L). Vanadium pore water chemistry is controlled by the redox behavior of vanadium in sediments. In reduced sediments, V is generally found in the highly insoluble V^{4+} form, not the more soluble V^{6+} valence (Fox and Doner 2003). For example, the low solubility of V^{4+} in sediments resulted in water vanadium in lagoon sediments with higher levels of total vanadium being a maximum of 45 µg/L and no V^{6+} being detected in either sediment or pore water (Nicholson et al. 2011). For these reasons, the analysis of aluminum and vanadium consisted only of comparisons of concentrations at individual sediment sampling locations to the REV.

4.1.2 Datasets Used to Evaluate Risk to Benthic Macroinvertebrates

To address each line of evidence for benthic macroinvertebrates, the following datasets are used:

- Concentrations of $COPC_Es$ in sediment (e.g., mg/kg) for each of the sediment samples collected from 0 to 15 cm (0 to 6 inches) for all aquatic portions of the Site. Where the elevation of the western cell is above the mean high tide, sediment samples are not included in the benthic invertebrate risk evaluation. This area is only occasionally inundated and does not provide appropriate habitat for a benthic macroinvertebrate community.

- Concentrations of TCDD (ng/kg ww) in individual clam samples collected from on the Site. Locations of transects at which clams were collected is shown in Figure 4-1.

4.1.3 Results of the Benthic Macroinvertebrate Exposure Evaluation

Summary statistics for estimated porewater concentrations for those chemicals evaluated using this line of evidence are presented in Table 4-2; their magnitudes relative to benchmarks are shown for each surface sediment sampling location on the Site in figures described in the risk characterization section. Estimated concentrations of phenol in porewater are not presented because phenol was not detected in 16 of 18 samples, and was J- or UJ-qualified in the other two.

4.2 Exposure of Fish

Lines of evidence to evaluate risk to fish include one of the following for each $COPC_E$:

1. Comparison of $COPC_E$ concentrations in the prey of fish to a TRV expressed as a concentration in food.
2. Comparison of estimated concentrations of $COPC_E$s in surface water to literature-based TRVs or benchmarks expressed as a concentration in water.
3. Comparison of the concentrations of total PCBs, $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in tissue of whole fish to CTRs expressed as a concentration in whole fish.

Data are not available to evaluate all lines of evidence for each receptor–$COPC_E$ pair. Information to address at least one line of evidence for each $COPC_E$ is presented.

For the second line of evidence, concentrations of $COPC_E$s in surface water were estimated from surface area-weighted average concentrations (SWAC) in sediment using the methods described in the previous section.

4.2.1 $COPC_E$ Concentrations in Fish Diets

For the first line of evidence, evaluation of exposure of fish through food and sediment ingestion, concentrations of $COPC_E$s in each ingested medium (food and sediment) is compared to the TRVs expressed as dietary concentrations (mg/kg diet). Where multiple prey types and sediment may be ingested by a fish (e.g., small fish and invertebrates), a concentration in the overall material ingested (food and sediment) was calculated using the following algorithm:

$$[COPC]_{diet} = \sum f1[COPC]_1 + f2[COPC]_2 \ldots + fn[COPC]_n \quad \text{(Eq. 4-4)}$$

Where:

$[COPC]_{diet}$ = concentration of COPC in the overall diet (µg/kg dw)

$[COPC]_{1 \ldots n}$ = concentration of COPC in ingested items 1 through n (µg/kg dw). Ingested items include both biological tissue and incidentally ingested sediment, if any

$f_{1 \ldots n}$ = fraction of prey items 1 through n in the overall diet (unitless), based on mass, the sum of which does not exceed 1.

The result of this calculation is a concentration in material ingested by fish weighted according to the proportion of each material type in the fish diet (Table 4-3). The result is directly comparable to TRVs expressed as a dry weight concentration in food of fish.

To evaluate species-specific exposures for each of the fish receptor surrogates, information on the proportions of each prey type in their diets was compiled from the literature (Table 4-3). Where the literature reported a prey type for which no Site-specific tissue chemistry data are available, the fraction of the diet consisting of that prey type is added to the fraction of an ecologically similar aquatic prey type for which Site-specific chemistry data are available. In Table 4-3, there are two columns showing the proportion of each prey type used by the fish: the proportion of the diet for each prey type reported in the literature, and the modified proportion of different prey types that are used in the algorithm above. Those prey types not represented in the baseline dataset, such as terrestrial invertebrates, are thus reassigned to an aquatic animal category: molluscs, crustaceans, or fish. For fish that ingest aquatic plants, the plant portion of the diet reported in the literature is distributed evenly among the prey types for which data are available. In this way, the range of prey ingested by each fish receptor is evaluated on the basis of empirical information, and results are reasonably conservative. Together, the three fish receptor diets realistically reflect different feeding guilds: the killifish (omnivore), black drum (benthic invertivore), and southern flounder (benthic piscivore) (Table 3-9).

This method is used to characterize exposure of fish to metals, because reliable TRVs expressed as CTRs for metals are generally not appropriate (Meador et al. 2010). USEPA (2007c) cautions against the use of CTRs for assessment of risk to aquatic organisms from exposure to metals (with the exception of organometals such as tributyltin and methylmercury), unless a toxicologically valid residue-response relationship supports the use of the CTR threshold. Metals are sequestered by many aquatic animals, and metals CTRs for fish are generally not reliable (Meador et al. 2011).

4.2.2 Estimated Concentrations of Selected COPC$_E$s in Surface Water

A TRV expressed as concentrations of bis(2-ethylhexyl)phthalate (BEHP), and of nickel in foods of fish were not found, so surface water concentrations of these two COPC$_E$s were estimated, and compared to TRVs. The approach used to estimate BEHP and nickel in water is analogous to the method to estimate porewater concentrations of some metals for the benthic invertebrate risk assessment (Section 4.1.1). Representation of surface water concentrations using an equilibrium partitioning approach is highly conservative, because the surface water has less direct contact with sediment than porewater, and is more dynamic making it less likely to reach equilibrium with sediment. Surface water is also diluted with instream and tidal flows. Regardless, using equilibrium partitioning methods to estimate surface water chemistry (as well as porewater chemistry) is a simplification of the aquatic and sediment environments, and the result is a highly conservative representation of water chemistry.

For this evaluation, the Site-wide concentration of selected COPC$_E$s is required, and the sediment concentration used is the SWAC.[4] To calculate the SWAC values for each COPC$_E$ requiring a water estimate, a set of Thiessen polygons was created using data from 0 to

[4] In addition to surface water concentrations for nickel, surface water concentrations of several other COPC$_E$s were estimated for use in the wildlife exposure model.

12 inches (including smaller increments such as 0 to 6 inch grab samples) below surface from each sediment sample location within USEPA's preliminary Site perimeter. The polygons were fitted to the perimeter boundary, and the area of each polygon and the total area were calculated. The concentration of the $COPC_E$ in each sample (in mg/kg dw for metals and in mg/kg OC for organic compounds) was then multiplied by the sample's corresponding Thiessen polygon area divided by the total area of the Site, or the fraction of the Site area represented by that sample. The sum of these surface area-weighted concentrations is the total SWAC for the $COPC_E$. The SWAC was used to represent C_S in Equations 4-1 and 4-2 (Section 4.1.1), producing an estimate of the overall surface water concentration on the Site. This result was used to evaluate exposure of fish to BEHP and nickel in the water, and to evaluate $COPC_E$ exposures through ingestion of water to wildlife, as necessary (below). $COPC_E$ SWACs are shown in Table 4-4.

4.2.3 Concentrations of PCBs and Dioxin-Like Compounds in Whole Fish

The third line of evidence is comparison of concentrations of total PCBs, $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole bodies of fish to TRVs expressed in the same terms (µg/kg ww or ng/kg lipid weight [lw]). Site-specific data are used for evaluating exposure of fish to dioxins, furans, and PCBs using these metrics. Composite whole killifish samples were collected on the Site from a series of transects, and 10 whole hardhead catfish samples were collected across all three fish collection areas (FCAs) on the Site (Figure 4-1), with four composites collected in FCA 2, the location of the northern impoundments. Background samples of Gulf killifish were collected from upstream of the Site (Figure 4-2), and background samples of whole hardhead catfish were collected only from Cedar Bayou (Figure 4-3).

Total PCBs in whole killifish and whole catfish from the Site were calculated as the sum of all 209 congeners, with the nondetects substituted at one-half the detection limit. The total PCB concentration in each fish sample as µg/kg ww is shown in Table 4-5 for each individual sample on the Site and for background. The wet weight concentration is used because it is compatible with available toxicity information (Appendix B).

Exposure of fish to dioxin-like compounds is expressed in a manner compatible with relevant toxicity information presented by Steevens et al. (2005): as the probability distribution of lipid-normalized, whole-body TEQ concentrations in Gulf killifish and hardhead catfish. TEQ_F concentrations are expressed as the $TEQ_{DF,F}$, or with $TEQ_{P,F}$ added ($TEQ_{DFP,F}$) in ng/kg lw. Steevens et al.'s (2005) SSD characterizes the distribution of threshold effect levels from tests with fish early life stages using exposures characterized as concentrations in eggs, developing young and maternal whole body of multiple fish species exposed to TCDD (Steevens et al. 2005; Appendix B). Use of all three exposure metrics is appropriate and may be conservative. A study of maternal transfer of TCDD in brook trout established a ratio of 0.39 between whole body and egg concentrations (Tietge et al. 1998). Studies of maternal transfer of other non-polar organic compounds support an approximately 1:1 egg to adult fish ratio (Russell et al. 1999).

Concentrations of $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole Gulf killifish and in whole hardhead catfish from each FCA on the Site and from background areas are shown for each sample in Table 4-6. Probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole Gulf killifish are shown in Figures 4-4 and 4-5, respectively, and probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole catfish are shown on Figures 4-6 and 4-7, respectively.

4.2.4 Unit Conversions

The following computations were used to convert data reported by the laboratory as wet weight to either a dry weight concentration (for calculation of prey concentrations for fish) or to a lipid weight concentration:

- To convert between concentrations expressed as wet weight and dry weight for tissue: mg $COPC_E$/kg dw = mg $COPC_E$/kg ww ÷ (1 – fractional moisture content)
- To convert concentrations expressed as wet weight to lipid-normalized concentrations: mg $COPC_E$/kg lipid = mg $COPC_E$/kg ww ÷ fractional lipid content.

Before calculating EPCs for tissue on a dry weight basis, wet-weight concentrations in individual samples were first converted to dry-weight concentrations using the fractional solids data (i.e., 1 – fractional moisture content) for the same sample if available; if solids data were not available, the average fraction of solids data for the given species was applied for the conversion.

4.2.5 Datasets Used to Evaluate Exposure to Fish

Fish were sampled from the three FCAs on the Site (Figure 4-8) and from Cedar Bayou (Figure 4-9) to represent background. On the Site, composite whole catfish samples were generated in FCA 1 (3 samples) FCA 2 (4 samples) and FCA 3 (3 samples) for a total of 10 whole catfish samples. In Cedar Bayou, 8 composite whole catfish samples were collected. Crabs were also sampled from the three FCAs on Site (Figure 4-10), and from Cedar Bayou (Figure 4-11). On the Site, composite whole crab samples were generated in FCA 1 (3 samples) FCA 2 (3 samples) and FCA 3 (3 samples) for a total of 9 whole blue crab samples. In Cedar Bayou, 3 composite whole blue crab samples were collected. Except for the association with a specific FCA, crab and catfish composites were not spatially referenced; each composite was created from samples across the FCA. Killifish and clam tissues were collected along transects (Figure 4-1), with composites representing the entire length of the transect. Additional information on tissue sampling is provided in the Tissue FSR (Integral 2011e).

In addition to the use of whole fish TEQ_F concentrations used noted above, the following data were used to perform the exposure evaluation for fish:

- **$COPC_{ES}$ in sediment (e.g., mg/kg) for sediment samples collected from 0 to 15 cm (0 to 6 inches) for all aquatic portions of the Site.** This dataset was used for estimating exposure to black drum and southern flounder, which would be expected to move around the entire Site and therefore be exposed to sediments throughout the Site.
- **$COPC_{ES}$ in sediment (e.g., mg/kg) associated with transects used for collection of Gulf killifish.** Killifish primarily move and forage in shallow nearshore intertidal habitats or inundated marsh surface habitats (Lotrich 1975). Home ranges have been estimated from 36 m to 0.15 km^2 across a marsh surface at low tide (Lotrich 1975; Teo-Able 2003). Most movements within tidal creeks in a mid-Atlantic estuary were within 200 m, with the majority of recaptures in a release-recapture study occurring within 50 m (Teo-Able 2003). An intermediate distance of 75m was selected to create a buffer around fish collection transects for selecting sediments to include in exposure assessment of Gulf killifish on a transect-specific basis. Buffers for Transects 1 and 2 largely overlapped and so were combined for determining a sediment dataset.

For calculating the prey-weighted concentrations in the diets of fish, the following were used:

- **$COPC_{ES}$ in whole crabs from the Site.** $COPC_{ES}$ were evaluated in crab tissue from the Site as a whole for drum and flounder, and using the nearest FCA for the transect-specific evaluation of exposure to Gulf killifish (crabs from FCA 1 were used to evaluate exposure to killifish in Transects 1 and 2, crabs from FCA 2 were used to evaluate exposure to killifish in Transects 3, 4, and 5, and crabs from FCA 3 were used to evaluate exposure to killifish in Transect 6).
- **$COPC_{ES}$ in edible tissue of clams from the Site.** Analogous to the approach for sediment, $COPC_{ES}$ in clam tissue from the Site as a whole were used in exposure calculations for drum and flounder, and on a transect-specific basis for the evaluation of exposure to Gulf killifish.
- **$COPC_{ES}$ in whole killifish from the Site.** Analogous to the approach for sediment and clam tissue, $COPC_{ES}$ in killifish from the Site as a whole were used in exposure calculations for drum and flounder, and on a transect-specific basis for the evaluation of exposure to Gulf killifish.

Exposure of fish to PCBs, dioxins, and furans was evaluated using empirical data on concentrations in whole catfish and killifish. Concentrations in individual samples used from Site and background are shown in Tables 4-5 for total PCBs and in Table 4-6 for dioxins and furans as $TEQ_{DF,F}$ and for dioxins, furans and PCBs as $TEQ_{DFP,F}$.

4.2.6 Results of Fish Exposure Assessment

Results of the exposure assessment for fish include:

- Weighted concentrations of the metals in fish diets (Table 4-7)
- Total PCB concentrations in individual whole fish samples (Table 4-5).
- TEQ concentrations in fish whole bodies for each sample (Table 4-6)
- Probability density functions for $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ in whole fish (Figures 4-4 through 4-7).

4.3 Exposure of Reptiles, Mammals, and Birds

Two lines of evidence are used to evaluate risks to birds:

1. Calculation of an individual's cumulative daily ingested dose of each $COPC_E$, expressed as mg $COPC_E$/kg body weight (bw) per day (mg/kg-day), and comparison of this estimate to a TRV expressed in the same terms.
2. Calculation of the $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentration in bird eggs (ng/kg egg, ww), and comparison to TRVs expressed in the same terms.

The first line of evidence is also used to address all $COPC_E$s in reptiles and mammals. Ingested media may include plant and animal foods, water, and soil or sediment as appropriate for each receptor's diet. The cumulative daily dose of each $COPC_E$ to an individual through ingestion of all relevant media each day was calculated using a wildlife exposure model (Section 4.3.1).

Application of this model requires designation of exposure units for each of these receptor groups, and calculation of EPCs within each exposure unit for use in the ingestion model. In addition, estimated concentrations of $COPC_E$s in foods for which empirical data were not available was required for the marsh rice rat and killdeer (exposure via ingestion of terrestrial invertebrates) and for raccoon (exposure via ingestion of plants and terrestrial invertebrates) (Table 4-8).

The second line of evidence is used to evaluate risk to the blue heron, neotropic cormorant, and spotted sandpiper from exposures to dioxins, furans, and dioxin-like PCB congeners. Modeling was used to estimate $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentrations in eggs, and their sum, from concentrations in media ingested by these birds (foods and sediment). Methods for this calculation are also presented below. This evaluation was not performed for killdeer because Site-specific empirical data were not available for the foods of killdeer.

The following sections describe the wildlife exposure model and algorithms necessary for making unit conversions, address the designation of exposure units, describe methods for estimating tissue concentrations as necessary and methods for calculation of EPCs to be used in the wildlife exposure model, and describes the models and assumptions used to estimate $TEQ_{DF,B}$ and $TEQ_{P,B}$ concentrations in bird eggs.

4.3.1 Wildlife Exposure Model

To estimate the cumulative daily dose for reptiles, mammals, and birds through ingestion of food and water, including incidental soil or sediment ingestion, the following general equation was used:

$$\text{Daily Dose} = \left((\text{FIR} \times C_{food} \times \text{RBA}_{food}) + (\text{WIR} \times C_{water}) + (\text{SIR} \times C_{sed} \times \text{RBA}_{sed}) \right) \times \text{AUF} \quad \text{(Eq. 4-5)}$$

Where:

Daily Dose	=	$COPC_{ES}$ ingested per day via food, water, and sediment (mg/kg bw-day)
FIR	=	food ingestion rate (kg food dw/kg bw-day)
C_{food}	=	concentration in the overall diet (mg/kg food dw)
RBA_{food}	=	bioavailable fraction absorbed from ingested prey items (unitless)
WIR	=	water ingestion rate (L water/kg bw-day)
C_{water}	=	concentration in water (mg/L water)
SIR	=	Sediment or soil ingestion rate (kg sediment dw/kg bw-day)
C_{sed}	=	concentration in sediment or soil (mg/kg dw)
RBA_{sed}	=	bioavailable fraction absorbed from ingested sediment or soil (unitless)
AUF	=	area use factor (unitless); fraction of time that a receptor spends at the Site relative to the entire home range.

Given that surface waters of the Site are brackish, wildlife other than seabirds and aquatic reptiles are not expected to ingest surface water at the Site, and the WIR term is set to zero for these other receptors. For those estuarine and marine receptors that could ingest Site water (great blue heron, neotropic cormorant, spotted sandpiper, and alligator snapping turtle), a WIR is provided based on allometric equations in USEPA (1993) and included in the exposure algorithm (Table 3-12).

Estimated values for those exposure parameters pertaining to receptor life histories were identified for each species using data compiled in the USEPA's Wildlife Exposure Factors Handbook (USEPA 1993) and other ERAs conducted within USEPA Region 6. Food ingestion rates were estimated using allometric equations presented in Nagy (2001) and USEPA (1993). Receptor profiles detailing the life history information providing the basis for wildlife exposure assumptions are provided in Appendix A. A summary of the selected exposure assumptions for use in the wildlife exposure model is presented in Table 4-8.

4.3.1.1 Ingestion of Multiple Prey Types

For those receptors likely to eat more than one prey type, the portion of the dose derived from the diet incorporates the proportion of each prey type within a typical diet for that receptor. This was done by weighting the $COPC_E$ concentration in each component of the diet by the fraction of the total diet consisting of that prey type. For example, the concentration of a $COPC_E$ in the diet of a receptor which ingests fish, benthic invertebrates and plants is estimated as follows:

$$C_{food} = (C_f \times F_f) + (C_i \times F_i) + (C_p \times F_p) \qquad \text{(Eq. 4-6)}$$

Where:

- C_{food} = concentration of the $COPC_E$ in the overall diet (mg $COPC_E$/kg food dw)
- C_f = concentration in fish tissue (mg COPC/kg tissue dw), where
  - C_{fs} = concentration in small fish tissue (e.g., killifish)
  - C_{fl} = concentration in large fish tissue (e.g., catfish)
- F_f = fraction of the diet consisting of fish (kg fish/kg food), where F_{fs} and F_{fl} are used to denote fractions of small and large fish in the diet, analogous to C_{fs} and C_{fl} above
- C_i = concentration in invertebrate tissue (mg $COPC_E$/kg tissue dw), where
  - C_{ic} = concentration in crustacean tissue (e.g., blue crabs)
  - C_{im} = concentration in mollusc tissue (e.g., common rangia)
  - C_{it} = concentration in terrestrial invertebrate tissue
- F_i = fraction of the diet consisting of invertebrates (kg invertebrates/kg food), where F_{ic}, F_{im}, and F_{it} are used to denote the fractions of various types of invertebrate tissue in the diet, analogous to C_{ic}, C_{im}, and C_{it} above

C_p = concentration in plant tissue (mg $COPC_E$/kg tissue dw)
F_p = fraction of the diet consisting of plants (kg plants/kg food).

Receptor-specific assumptions about the fraction of each prey type in the diet and the information sources supporting each assumption are shown in Table 3-12. It is recognized that individuals of these receptor species may ingest prey types in proportions different from that shown here. The proportions of each type of food in the diet of any given receptor are intended to broadly represent the receptor and its feeding guild.

4.3.1.2 Relative Bioavailability Adjustment Factor

Except for the calculation of daily ingestion rates of dioxins and furans by birds, wildlife exposure calculations for all $COPC_E$s conservatively assume that bioavailability in all media ingested in the field is the same as in the laboratory toxicity study that provides the basis for the TRV. This is a conservative assumption, because laboratory toxicity studies are often conducted using a highly soluble or dissolved form of the chemical in water or food, while the exposure in the field is to the chemical bound in a particular matrix (e.g., food or sediment), in multiple compartments within ingested prey species (such as muscle, bones, and blood), or is otherwise in a form that is not analogous to the laboratory exposure. However, given the variety of mechanisms used to administer test substances in laboratory studies, it is not appropriate to apply a single adjustment factor for all aspects of exposure and all chemicals. In the absence of compelling information for individual chemicals in specific ingested media, this exposure assessment does not apply relative bioavailability adjustment (RBA_{food} and RBA_s) factors in the wildlife exposure model.

One study was found to support application of an RBA in calculation of ingestion exposure by birds. Nosek et al. (1992a) tested the oral bioavailability of TCDD to adult pheasant hens. They mixed radiolabeled TCDD into a suspension of worms, a suspension of crickets, a suspension of paper mill sludge, and a suspension of soil, and administered a fixed amount of the chemical in each suspension into the crops of tested bird in a single dose. After dosing, the birds were allowed to eat normal feed *ad libitum*. After 24 hours, birds were sacrificed, the entire digestive tract removed, and the radioactivity remaining in the bird carcass was measured. Nosek et al. (1992a) report the following absorption rates from the different materials tested: earthworms, 30 percent; soil, 33 percent; paper mill solids 41 percent; and crickets, 58 percent. These were used to derive RBA factors for foods and soil or sediment.

A separate study by Nosek et al. (1992b) provides the basis for derivation of the TRV used in this BERA to interpret daily ingestion rates of dioxins and furans. In that study, adult pheasant hens were administered specific doses of TCDD via intraperitoneal injection. To derive a TRV, the weekly injections were converted to a daily rate and used as an approximation of the daily ingestion rate (Appendix B). While providing a very precise measure of dose, using intraperitoneal injection as the basis for an estimate of an ingestion rate is highly conservative, representing an assumption of 100 percent absorption of TCDD in ingested media through the gut.

In addition to the evidence provided by the robust and systematic analysis by Nosek et al. (1992a), there is a substantial body of evidence to support the assumption that oral bioavailability of TCDD is less than 100 percent in most (and possibly all) vertebrates. Limitations on uptake of all dioxin and furan congeners can be explained by both biological factors and physicochemistry of dioxins and furans in abiotic environmental exposure media. Opperhuizen and Sijm (1990) postulated that the relatively large size of dioxin molecules limits uptake across gill and gut membranes in fish, and that the limitation on uptake rates for dioxin and furan congeners increases with increasing chlorination, which corresponds to molecular size. This conceptual model was confirmed by evaluation of several independent lines of evidence in an analysis for this project (Integral 2010b). Moreover, USEPA (2010b) recently summarized several experimental studies with mammals demonstrating limited uptake of dioxins and furans from weathered soils (i.e., soils contaminated in an environmental context, not spiked for the test). They conclude that although there is variability among species, there is substantial evidence of limited oral bioavailability of dioxins and furans. Budinsky et al. (2008) provide an excellent review of the literature in their introduction, citing a range of experimental data on both limited absorption of dioxins and furans in mammalian gastrointestinal systems, but also limited desorption from ingested media within the gut, an additional factor controlling uptake independent of membrane pore size.

Although the experimental data for mammals and fish, as well as Nosek et al.'s (1992a) bioavailability data, conform to the conceptual framework advanced by Opperhuizen and Sijm (1990) and supported by Budinsky et al. (2008), experimental studies with birds are limited. In one study, Stephens et al. (1995) mixed clean soil, soil contaminated in the field with all 17 dioxin and furan congeners, and dioxin- and furan-spiked soil into the feed of chickens and periodically monitored concentrations of all congeners in various tissue types. Tissues were sampled during an exposure period of 178 days and during depuration of another 100 days. They used a mass balance approach to estimate overall uptake and retention during this experiment. They find that for all congeners, less than 100 percent of mass could be accounted for by their mass balance, and attribute this limited mass to constraints on bioavailability. Although likely correct in this interpretation, their results do not provide the basis for quantitative estimates of RBA because both their dose estimates and their mass balance calculations were imprecise. Nonetheless, they report that after 164 days, less than 65 percent of the ingested TCDD was present in tissues of chickens. This result was comparable to that for TCDF, and all other congeners were present at even lesser percentages of the total mass ingested by the chickens. In addition to this study, Nosek et al. (1992a) cite a study by Martin et al (1989), in which European starlings were dosed with radiolabeled TCDD in an experiment similar to that of Nosek et al. (1992a). Martin et al. (1989) tested oral bioavailability in starlings from suspensions of earthworms, paper mill sludge, soft-bodied invertebrates, and hard-bodied invertebrates, finding bioavailability of TCDD in these suspensions to be 14, 17, 37, and 44 percent, respectively. These results are generally consistent with the RBAs used in this risk assessment, but they indicate even more attenuated uptake in the guts of the starling than the pheasant. However, this paper was published in the grey literature and Nosek et al.'s (1992a) interpretation could not be independently confirmed.

Overall, the weight of evidence presented by Opperhuizen and Sijm (1990), USEPA (2010), Budinsky et al. (2008), Stephens et al. (1995) and Nosek et al. (1992a) clearly supports application of a bioavailability adjustment factor for TCDD. The values presented by Nosek et al. (1992a) and used in this risk assessment are technically robust and appropriately conservative.

To use the media-specific information on relative bioavailability provided by Nosek et al. (1992a), RBA factors for 2,3,7,8-TCDD in soil, sediment and invertebrate tissue were derived (Table 4-9). The RBA factor for invertebrates is the average of the two absorption rates reported for earthworms and crickets (30 percent and 58 percent, respectively), or 0.44. Nosek et al.'s (1992a) result for paper mill sludge is used as the RBA for sediment, and the result for soil is used for soils. For the wildlife exposure model, the 2,3,7,8-TCDD concentration was multiplied by the medium-specific RBA factor prior to calculation of the TEQ for this congener. For terrestrial invertebrate tissue concentrations estimated from soils, the RBA was multiplied by the concentration of 2,3,7,8-TCDD estimated for tissue from a regression relationship (Appendix D). The resulting adjusted TCDD concentration was then multiplied by the TEF to calculate the adjusted TEQ for TCDD in tissue, and added to TEQ concentrations for the other congeners within the sample to calculate a $TEQ_{DF,B}$ or $TEQ_{DFP,B}$. To evaluate the effect of this adjustment on the risk calculations, a sensitivity analysis was conducted using TEQs which were calculated without the application of the RBA for 2,3,7,8-TCDD. The results of this sensitivity analysis are discussed in the uncertainty evaluation (Section 7).

4.3.1.3 Unit Conversions

It is conventional for laboratories to report analytical results for tissue in wet weight concentrations. However, total food ingestion rates, which form the basis for the wildlife exposure model, are estimated on the basis of energy requirements, which are computed from the dry mass of different food types. To convert concentrations expressed as wet weight to dry weight concentrations for tissue, the following equation is used:

$$\text{mg COPC}_E/\text{kg dw} = \text{mg COPC}_E/\text{kg ww} \div (1 - \text{fractional moisture content}) \quad \text{(Eq. 4-7)}$$

Before calculating EPCs for tissue on a dry weight basis, wet weight concentrations in individual samples are first converted to dry weight concentrations using the fractional solids data for the same sample if available; if solids data is not available, the average fraction of solids data for the given species is used.

4.3.1.4 Estimation of Concentrations in Whole Fish

Ecological receptors that eat fish are assumed to ingest the entire fish. Whole hardhead catfish and whole crabs were not sampled for the RI, but remainder samples were collected with samples of edible tissue from both species. Tissue masses of both edible and remainder tissues for catfish and crab were measured in the laboratory. For each individual sample, a mass-weighted whole fish concentration was calculated as follows:

$$C_{wb} = \left[C_{fi} \times \frac{W_{fi}}{W_{wb}}\right] + \left[C_r \times \frac{W_r}{W_{wb}}\right] \quad \text{(Eq. 4-8)}$$

Where:

C_{wb}	=	chemical concentration in the whole body of the fish (mg/kg)
C_{fi}	=	chemical concentration in the fillet tissue (mg/kg)
C_r	=	chemical concentration in the remainder tissue (mg/kg)
W_{fi}	=	weight of the fillet (kg)
W_{wb}	=	weight of the whole body (kg)
W_r	=	weight of the remainder (kg).

Resulting “whole fish” concentrations were the only fish and crab tissue data used for the BERA; no calculations were performed with just edible crab or just catfish fillet.

Gulf killifish and clams were collected whole. Clams were briefly held in buckets after sampling, allowing them to excrete any sediment in their gut prior to chemical analysis. All soft tissues were extracted from the clam (everything inside the shell) for analysis, and it was these soft tissues that were used for the BERA.

4.3.1.5 Concentrations of $COPC_E$s in Foods of Alligator Snapping Turtle, Killdeer, Raccoon, and Marsh Rice Rat

Empirical data are available to describe concentrations of $COPC_E$s in soils, sediments, fish and aquatic invertebrate tissue from the Site and from background areas. Methods for calculation of EPCs for these media, for use in the wildlife exposure model, are presented in Section 4.3.1.7, below. There are no empirical data to describe $COPC_E$ concentrations in terrestrial invertebrates and plants from the Site. As a result, it was necessary to estimate $COPC_E$ concentrations in terrestrial invertebrates from soil concentrations, and to estimate

$COPC_E$ concentrations in aquatic plants from sediment concentrations and in terrestrial plants from soil concentrations. To do this, simple bioaccumulation factors or regression equations describing relationships between concentrations in sediment or soil and the concentrations in invertebrate prey or plant tissue were used. For the majority of $COPC_E$s, these equations take the following forms:

$$C_{prey} = C_s \times BAF_{prey} \qquad \text{(Eq. 4-9)}$$

$$C_{prey} = B1_{prey} \times C_s + B0_{prey} \qquad \text{(Eq. 4-10)}$$

Or

$$\ln(C_{prey}) = B1_{prey}(\ln[C_s]) + B0_{prey} \qquad \text{(Eq. 4-11)}$$

Where, for those chemicals lacking a statistically significant regression model:

BAF_{prey} = bioaccumulation factor (kg dw soil/kg dw prey)

And for those chemicals for which a significant regression model is available:

$B1_{prey}$ = slope of the regression of the concentration of the chemical in the prey against the concentration of the chemical in soil or sediment

$B0_{prey}$ = *y*-intercept term, describing the concentration in the prey when $C_s = 0$

The BAF is simply the ratio of the concentration of chemical in the prey (C_{prey}) to the concentration of the chemical in sediment or soil (Cs). As discussed by Integral (2011), regression models have several technical advantages over simple ratios, and are considered the most appropriate method for analysis and characterization of relationships between abiotic media and tissue. Generally, regression models were preferred for making the predictions of chemical concentrations in tissue necessary for wildlife exposure modeling, consistent with Integral (2011).

For chemicals other than dioxins and furans, the primary source of models for this analysis was USEPA's Guidance for Developing Ecological Soil Screening Levels (USEPA 2007c), which provides regression equations describing relationships between measured concentrations of chemicals in soils and plants, and between concentrations in soil and terrestrial invertebrates. When BAFs or regression equations were not available from USEPA

(2007c), other sources were reviewed, including Sample et al. (1998), USEPA (1999b), and RAIS (2011). Staples et al. (1997) describe the environmental chemistry of BEHP, and provided the relevant information for predicting BEHP in tissue. Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations of mercury in terrestrial invertebrates from Site soils. For dioxins and furans, no model was selected for estimating concentrations in plants. A dataset for a Superfund site in Minnesota was analyzed to derive regression models for estimating concentrations of individual congeners in terrestrial invertebrates, as described below.

4.3.1.5.1 Estimating $COPC_E$ Concentrations in Plants

To estimate metals concentrations in plant tissue, BAFs or regression models from USEPA (2007c) were available for cadmium, chromium, cobalt, copper, lead, nickel, vanadium, and zinc (Table 4-10). Plant BAFs were not available from USEPA (2007c) for dioxins and furans, PCBs, BEHP, or mercury. For mercury, soil-to-plant BAFs were selected from USEPA (1999b). The BAF for mercury is the recommended BAF for mercuric chloride. Results of this evaluation are not presented in tabulated form.

For dioxins and furans, PCBs and BEHP, uptake from sediments into plants is considered negligible (Wild and Jones 1992; Bromilow and Chamberlain 1995; Staples et al. 1997). Plants are exposed to $COPC_{ES}$ in sediment primarily through porewater (Kabata-Pendias and Pendias 1992). Chemicals with low water solubility may adsorb to the roots of plants (e.g., lead), but uptake into the plant's vascular system and transport to leaves and fruit is limited for low-solubility chemicals. Lipophilic compounds are taken up by the roots or by foliage, and are transported in plant xylem. This transport is slowed by partitioning of lipophilic compounds to the lipid-like matter in plant tissue (Bromilow and Chamberlain 1995), and fruits are not affected. If taken up by plants, lipophilic chemicals tend to accumulate in the leaf margins and interveinal spaces. As a general indicator of the transport of lipophilic compounds within plants, Travis and Arms (1988) reviewed BAFs for plant foliage for 29 chemicals with log K_{ow} values ranging from 1 to 10, and found that the BAF was inversely proportional to the square root of the K_{ow}.

Although USEPA (1999b) provides a relationship for BEHP based on its K_{ow}, these types of simplified relationships based solely on chemical hydrophobicity are limited because they do not take into account processes such as metabolism which play an important role in modulating phthalate bioaccumulation (Staples et al. 1997). Staples indicates that most soil-to-plant BAFs for phthalates are <0.1 and typical soil-to-plant BAFs from their literature review are <0.01. Staples et al. (1997) also note that these BAF values are based on studies with radiolabeled carbon, and exposures to multiple phthalates. As such, resulting BAFs overestimate final tissue concentrations of the phthalate because metabolites in tissues are not distinguishable from the parent compound using this method. Other studies discussed by Staples et al. (1997) suggest there is no appreciable uptake of BEHP to plants from soils.

Much of the data concerning plant-uptake of organic chemicals from soil comes from studies investigating soil-to-plant transfers of chemicals derived from sludge-amended soils. Regarding these studies, Wild and Jones (1992) provide the following general comments including PCBs, polycyclic aromatic hydrocarbons (PAHs), and other organochlorine compounds typically studied, “these compounds are generally not taken up into the aboveground portion of crop plants” and, that “there is some evidence of slight enrichment of some compounds in some root crops, but the transfers are very inefficient, and consequently the BCFs [bioconcentration factors] are very low.” They also note that enrichments are generally confined to the root skin or peels, and not the fruits, leaves, and stems that may be eaten by wildlife (Wild and Jones 1992).

For these reasons, plant tissue concentrations of dioxins, furans, PCBs, and BEHP are assumed to be zero in the wildlife exposure model.

4.3.1.5.2 Estimating $COPC_E$ Concentrations in Soil Invertebrates

BAFs to estimate terrestrial invertebrate tissue concentrations were available from USEPA (2007c) for cadmium, chromium, cobalt, copper, lead, vanadium, and zinc (Table 4-10). Soil-to-invertebrate BAFs were available for nickel in USEPA (1999b). For PCBs, a regression equation from Sample et al. (1998) was selected to estimate total PCB concentrations in soil invertebrates from total PCB concentrations in soil. Congener-specific models were not used because there are no PCB congener data for soils at the Site with the exception data for soils collected from the Texas Department of Transportation right-of-way.

Sample et al. (1998) and other compendia do not provide robust regression relationships for mercury, so Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations from Site soils. Burton et al. (2006) conducted a 28-day study evaluating uptake by earthworms of total and organic mercury from soils across a range of mercury concentrations. Their study established that mercury concentrations in earthworm tissue were higher relative to concentrations in soils at lower soil concentrations, and lower relative to concentrations in soils at higher soil concentrations. Burton et al. (2006) report a BAF of 3.1 for low soil concentrations (0.156 mg/kg) and a BAF of 0.6 and 0.7 for intermediate (2.83 mg/kg) and high (11.54 mg/kg) soil concentrations. The method to estimate mercury concentrations in terrestrial invertebrate tissue for the wildlife exposure model recognizes this differential in mercury BAF with concentration in soil because mercury concentrations in soils are either less than 2 mg/kg or greater than 10 mg/kg (Figure 4-12).

Burton et al. (2006) did not establish a measure of variance around their surface soil concentrations, so the division between low and intermediate concentrations was established as the median value between 0.156 and 2.83 mg/kg, or 1.5 mg/kg. Therefore, to estimate concentrations of mercury in terrestrial invertebrates for the wildlife exposure model, the BAF of 3.1 was applied at stations with mercury concentrations below 1.5 mg/kg, and the BAF of 0.7 was applied to Site surface soils with concentrations greater than 1.5 mg/kg. Burton et al.'s (2006) BAFs for intermediate and high soil concentrations were not significantly different, so the choice of the higher BAF of 0.7 for estimating tissue concentrations at the stations with mercury concentrations in soil greater than 1.5 mg/kg was conservative (Figure 4-12).

At least one recent study both supports the use of this approach and illustrates that the use of a BAF simplifies a likely complex system. Fengxiang et al. (2012) reports on earthworm bioaccumulation studies using soils with and without cinnabar mercury, and with mature and immature worms. They report that cinnabar mercury, tightly bound to sulfur, does not correlate with mercury in worm tissue, while non-cinnabar mercury correlates well. This example illustrates the importance of local soil conditions. Fengxiang et al.'s (2012) soil-to-earthworm BAFs for mercury ranged from 0.32 to 1.75, indicating that the BAFs selected for this BERA are conservative (i.e., above the upper end of that range for uptake from soils with low concentrations of mercury).

Staples et al. (1997) presents information to indicate that BEHP does not bioaccumulate in soil invertebrate tissue at environmentally realistic concentrations in soil. Only one study (Hu et al. 2005) reported biological transfer of BEHP from soils to soil invertebrates, but exposure concentrations were much higher than the range of BEHP concentrations in Site soils. Therefore, the reported soil-to-invertebrate bioaccumulation relationship reported by Hu et al. (2005) is not appropriate for application to Site conditions. On the basis of the review provided by Staples et al. (1997), invertebrate tissue concentrations of BEHP are assumed to be zero in the wildlife exposure model.

None of the literature sources listed above provides sufficient information for use in estimating concentrations of dioxin and furan congeners in terrestrial invertebrates. Although Sample et al. (1998) present a model for TCDD, important details are missing from the analysis, and only one congener is addressed. To develop soil-to-invertebrate relationships for predicting dioxin and furan concentrations in tissue, data from a bioaccumulation study using earthworms (*Eisinia fetida*) at a Superfund site in Minnesota were used. In two locations from this study, naive earthworms were exposed to samples of field-collected soils contaminated with dioxins and furans for 28 days. At the end of the test, animals were purged and tissues analyzed for dioxins and furans. The study was conducted according to specifications of the American Society for Testing and Materials Method 1976–04. In an additional five locations, co-located soils and earthworm tissue were collected. One worm froze in transit and could not be depurated, and therefore was not used for developing soil-to-tissue relationships. All worms used from this dataset were depurated prior to analysis. All worm and co-located soils were analyzed for dioxins and furans using USEPA Method 8290. All chemistry data were validated according to CERCLA validation protocols. More information on this study, and the data used, are provided in Appendix D. Results include a series of regression and correlation relationships for dioxin and furan congeners, summarized in Table 4-11, that were used to estimate dioxin and furan concentrations in soil invertebrate tissue for use in the wildlife exposure model for killdeer and raccoon. Additional methodological details and results of statistical evaluations and resulting tissue concentration estimates are provided in Appendix D.

Soil-to-earthworm BAFs or regression relationships are summarized in Table 4-10.

4.3.1.6 Wildlife Exposure Units

An exposure unit is the area in which a receptor may be exposed to contaminants in environmental media. The exposure unit provides an organizing concept for selection of data to be used in estimating wildlife exposures. It defines the spatial area from which data were selected for calculation of EPCs for each medium, using samples collected according to the DQOs presented in relevant SAPs. An individual receptor is assumed to be equally likely to be exposed to media within all subareas of the exposure unit.

Wildlife exposure units for this BERA were defined to reflect the possible foraging areas and habitats for the surrogate receptor species at the Site (Appendix A). The exposure units for reptiles, birds, and mammals included the following areas of the Site:

- Upland habitat for evaluation of exposures to raccoon and killdeer within USEPA's preliminary Site perimeter, including soils and foods in:
  - All upland habitat north of I-10 (for killdeer, Figure 4-13)
  - All upland habitat on the peninsula within and adjacent to the impoundments, both north and south of I-10 (for raccoon, Figure 4-14)
- Shoreline habitat within USEPA's preliminary Site perimeter, including sediments, surface water (ingested by aquatic birds and reptiles) and prey within these exposure units (Figure 4-15)
- Aquatic habitat within USEPA's preliminary Site perimeter, including sediments, surface water (ingested by aquatic birds and reptiles) and prey within these exposure units (Figure 4-16 and 4-17).

When using the Site, a given receptor may be present in one or more of these exposure units, depending on its life history and foraging habits. Although receptors would use an area according to its habitat quality and resources provided by the habitat (forage, refugia), the approach used to establish EPCs conservatively assumes that receptors will be more likely to encounter contaminated areas than other areas on the Site, regardless of habitat quality. Concentrations in exposure media were not spatially weighted; instead, each sample was given equal weight, even though there is a higher spatial density of samples directly adjacent to the impoundments north of I-10 than elsewhere within USEPA's preliminary Site perimeter. Samples were not collected evenly across all habitats, such as the vegetated area to the west of the sand separation area, or the eastern shoreline. This spatial distribution of samples reflected DQOs described by approved SAPs.

Table 4-8 outlines the way in which exposure units and media are assigned to each receptor for the wildlife exposure assessment. Figures are presented to graphically illustrate the exposure units for each receptor surrogate, as follows:

- Figure 4-16, Alligator snapping turtle
- Figure 4-17, Neotropic cormorant
- Figure 4-15, Great blue heron, spotted sandpiper, and marsh rice rat
- Figure 4-14, Raccoon
- Figure 4-13, Killdeer.

Each of these figures shows the sediment and/or soil samples, and the transects for tissue collection where applicable, used for calculating EPCs for the estimate of exposure to each wildlife receptor. Because most samples were collected in locations near or adjacent to the impoundments north of I-10, regardless of habitat quality, and because all samples were given equal weight in exposure statistics, regardless of the spatial area represented, the selection and definition of exposure units was conservative. For the post-TRCA scenarios, all samples collected from within the original 1966 perimeter of the impoundments north of I-10 were removed from the data before performing calculations, and replaced with one value equal to the median concentration of the upstream background sediment or the background soil data, as appropriate, for the chemical of interest.

Data selected for calculating exposures in the aquatic environment were selected by clipping the hydrologic unit polygon for the San Jacinto River to the preliminary Site perimeter boundary. The hydrologic unit polygon was received from the Harris County Public Infrastructure Department Architecture and Engineering Division. This polygon was transformed into a line feature which was clipped appropriately and used to represent and calculate total length of the shoreline within the site boundary.

Data for calculating exposures in terrestrial areas were selected using digitized polygons based on 0.5-m 2008/2009 Digital Orthophoto Quarter Quads from the Texas Strategic Mapping Program (StratMap) that most closely represent the habitat of the organism of interest. Only soils collected from 0 to 6 inches depth were used. The habitat area calculations were used to estimate Site exposure unit sizes for each receptor (Table 4-12).

For protected species that could occur on the Site (white-faced ibis, bald eagle, and pelican), if the estimated exposure of their respective avian receptor surrogates (Section 3.3) to a $COPC_E$ exceeds the NOAEL or the LOAEL, then the exposure of each of the protected species to that chemical is calculated by adjusting the exposure area assumed for surrogate (as described above) by the relative size of the protected species' home range. Because the home range of each surrogate for the protected species that could occur on the Site is conservatively assumed to be equal to the exposure unit, this calculation consists of multiplying the dose by the ratio of the surrogate's exposure unit area to the protected species' home range area (Table 4-12). Results are addressed for relevant $COPC_{ES}$ in Section 6.

4.3.1.7 Calculation of Exposure Point Concentrations

Consistent with USEPA guidance (USEPA 1997) which directs ecological risk assessors to consider an exposure profile for each receptor, EPCs were generated for each exposure medium within each exposure unit for use in the wildlife exposure model described above. CT and RM exposure concentrations were generated for each $COPC_E$. Selection of the appropriate statistic to represent the CT and RM for each EPC was based on the statistical distribution of the data supporting that EPC for each $COPC_E$ within a given medium (sediment, soil, and tissue) and exposure unit. All analyses of data distributions and generation of distribution parameters were performed using the software R for Windows version 2.9.0 (R Development Core Team 2008).

Treatment of censored data in EPC calculations is discussed in Section 2.2.2. Decisions for generation of the statistical representations of the EPCs for a given data distributions were as follows (Appendix C):

- For normal data distributions, the arithmetic mean was chosen as the CT and the 95UCL based on a Gaussian data distribution was selected as the RM.

- For lognormal distributions, the geometric mean was chosen as the CT and the 95UCL based on a lognormal data distribution was selected as the RM.
- For unknown data distributions (i.e., those distributions that were not normal and could not be transformed to a log-normal distribution), the arithmetic mean was chosen as the CT and 95UCL was calculated using nonparametric statistics, consistent with ProUCL (USEPA 2007b).

In all cases, if the 95UCL was greater than the maximum value for the dataset, the maximum was selected as the RM. Results of all EPC calculations are presented in Appendix C.

For a few datasets (e.g., TEQ_P in soil and shoreline sediments), the sample size was so small ($N < 4$) that a distribution of the data could not be calculated and a UCL could not be generated with confidence; in these cases, the maximum value was used as an estimate of the RM. For a few other datasets (BEHP in clams and Gulf killifish), there were no detected values, so the CT and RM in these cases were set equal to one-half the detection limit. Concentrations of PCBs in water were not estimated, and the PCB doses via ingestion of water for seabirds were not calculated, because the dose via water ingestion is assumed to be minor relative to dose via ingestion of foods due to the low solubility and relatively high potential for bioaccumulation and biomagnification of PCBs.

To estimate concentrations of $COPC_{ES}$ other than dioxins and furans in terrestrial invertebrate and plant tissue, a soil or sediment EPC calculated using data from within the exposure unit of the subject receptor is multiplied by the BAFs or is used in the regression equations (Table 4-10) to generate CT and RM EPCs for input to the wildlife exposure model.

Where an analysis of the post-TCRA wildlife exposure is needed, all samples for stations within the original 1966 impoundment perimeter are removed and replaced with a single value representative of the possible post-TCRA condition. The value used in these substitutions is the median concentrations of the $COPC_E$ in the upstream sediment dataset or in the background soil dataset, depending on whether the exposure scenario involves exposures to sediments or soils. All of the analyses to describe the data distribution and to calculate CT and RM EPCs were repeated using this substituted dataset prior to their use in the wildlife exposure model. Results are presented in Appendix C. No substitutions were performed for tissue concentrations, so pre-TCRA tissue concentrations were used in post-TCRA analyses.

4.3.1.8 Data Used

The data used in the wildlife exposure model include:

- Sediment and soil samples collected from 0 to 6 inches shown in Figures 4-13 through 4-17
- Sediment from 0 to 6 inches from the upstream sediments background study
- Soil from 0 to 6 inches from the Site-specific background study
- All clam samples collected for the RI (Figure 4-1)
- All killifish samples collected for the RI (Figure 4-1)
- Whole hardhead catfish samples and whole blue crab samples from on the Site (Figure 4-1)
- Tissue samples collected from the upstream background (Figure 4-2) and Cedar Bayou (Figure 4-3) background tissue study
- Surface water samples collected by TCEQ for analysis of dioxins and furans (URS 2010).

Soil from the Site specific background study, sediment data from the upstream background area, and tissue data from background areas were used only when the $HQ_L \geq 1$ (Section 3.8).

4.3.1.9 Results

Summary presentations of results of the wildlife exposure model and supporting calculations are provided as follows:

- Results of calculations using BAFs and regression models for invertebrates and plants were not tabulated, but were incorporated directly into the wildlife exposure model
- The EPCs used in the ingestion model are presented in Appendix C, Table C1.
- Final estimates of the daily ingestion rate of each $COPC_E$ for each bird, mammal, and reptile receptor surrogate are shown in Table 4-13.

4.3.2 Estimated TEQ Concentrations in Bird Eggs

Concentrations of dioxin-like compounds in bird eggs were estimated as part of the exposure assessment because substantial toxicity information in the literature for birds is expressed as egg concentrations (RI/FS Work Plan, Appendix B, Attachment B2), and because comparison of TEQ concentrations in eggs to TRVs expressed as egg concentrations is the risk assessment method recommended by USEPA (2003b; 2008). Site-specific data to described TEQ concentrations in bird eggs were not developed for the RI, so modeling was performed to derive estimates of egg concentrations. Methods for modeling were different for dioxins and furans than for dioxin-like PCBs, due to differences in the information available in the literature. Each method is described below.

4.3.2.1 Estimating Dioxins and Furans in Bird Eggs

The uptake of dioxins and furans from dietary sources into bird tissue and subsequent transference into eggs is both species- and congener-specific. This process can be considered as occurring in two steps: 1) the uptake and retention of dioxins and furans by the egg laying female, and 2) the maternal transfer of dioxins and furans into the egg. Although uptake and retention of dioxin and furans in vertebrates is species- and congener-specific, general trends can be found in the literature (Integral 2010b). In contrast, maternal transfer of dioxins and furans from egg laying female birds to their eggs has been less well studied, and sufficient information for mechanistically modeling egg concentrations stepwise through these two process steps was not found. As a result, the simple bioaccumulation from foods ingested by the parent bird into eggs provides the conceptual basis for estimating egg concentrations for this evaluation.

Simple estimation methods such as biomagnification factors (BMFs) calculated as the ratio of a food concentration to an egg concentration, can lead to significant error in predicted egg tissue chemistry. This potential for error is due to congener- and species-specific differences in retention and distribution of dioxins and furans (Integral 2010b). If appropriate data are available, use of statistical regression models overcomes several weaknesses in the ratio method.

4.3.2.1.1 Identification of a Prey-to-Egg Regression Model

A literature search was conducted to identify studies describing statistical models of prey-to-egg relationships, and only one paper was found to support this analysis (Elliott et al. 2001). The study presented by Elliott et al. (2001) provides a set of regression models for estimating dioxin and furan concentrations in bird eggs from concentrations in foods of the birds. Elliott et al. (2001) focus on the great blue heron, a piscivore and one of the avian receptors at the Site. Congener-specific and homologue-based regression models reported by Elliott et al. (2001) for log-transformed dioxin and furan data in prey tissue were used to estimate egg tissue TEQ concentrations from dioxins and furans in ingested media from the Site and background areas, and to estimate post-TCRA exposures.

Elliott et al. (2001) monitored dioxins and furans in eggs from 21 great blue heron rookeries and in prey fish from 1983 to 1998. They developed linear regression models showing strong positive relationships between congener families and TCDD and TCDF in prey fish species and in heron egg tissue (Table 4-14). A review of the literature and subsequent reanalysis of published data showed linear relationships between diet and egg dioxin and furan concentrations from two other studies:

- **Tree swallows in Woonasquatucket River, Rhode Island (Custer et al. 2005):** Evaluation of data for this site by Integral indicated a linear relationship between dioxin and furan concentrations in pooled diet samples and egg tissue (when nondetects in diet samples are excluded).
- **Herring gulls in Lake Ontario (Braune and Nordstrom 1989):** Evaluation of data from this study indicated a moderate linear relationship between alewife prey and egg tissue concentrations of dioxins and furans.

Although both of these studies support the selected approach for modeling egg tissue concentrations, the data were insufficient for use in developing a model for egg tissue estimates. As a result, only the regression models reported by Elliott et al. (2001) were used. Results of the studies with herring gulls and swallows improve confidence in the conceptual basis for the selected approach.

4.3.2.1.2 Implementation of the Prey-to-Egg Model

The linear regression models for each congener or homologue group from Elliott et al. (2001) (Table 4-14) were used to estimate egg concentrations for three bird receptor surrogate species: blue heron, cormorant, and sandpiper. The independent variable used in each model, which was fish tissue concentration in Elliott et al. (2001), was estimated to reflect the aggregate of ingested media by the receptor surrogates for the Site. Ingestion of contaminated prey with and without ingestion of contaminated sediment was evaluated. Ingested media in this model for each bird receptor were as follows:

- **Blue heron:** Whole catfish, whole blue crab, Gulf killifish, and shoreline sediment (Figure 4-15)
- **Cormorant:** Gulf killifish and bottom sediment (Figure 4-17)
- **Sandpiper:** Clams, whole blue crabs, and shoreline sediment (Figure 4-15).

The regression models required individual congener or homologue data for each ingested medium. Using the CT and RM for each individual congener was considered overly conservative because to do so would result in a combination of dioxin and furan congeners for each ingested medium that would not be representative of the congener composition and TEQ concentrations in the natural environment and in actual tissue samples. Moreover, this approach would be inconsistent with exposure profiles represented by the CT and RM of TEQ elsewhere in this risk assessment. Instead, an individual sample of each medium was selected to represent the CT and RM exposures. To do this, the CT and RM $TEQ_{DF,B}$ concentrations of each medium within each exposure unit were calculated. Because the result is a statistic, and not a specific sample, the actual sample with the $TEQ_{DF,B}$ concentration closest to the CT and the sample with the $TEQ_{DF,B}$ concentration closest to the RM were identified. For each medium, the sample number and congener concentrations under each scenario are shown in Table 4-15. The physical locations of the samples in Table 4-15 are referenced in Figures 4-15 and 4-17. The specific congener concentrations within these samples were used in subsequent calculations. Examples of several specific calculations for the cormorant, heron, and sandpiper are presented in Exhibits 2A and 2B.

To estimate the congener or homologue concentrations in eggs that accounted for all ingested media, a mass-weighted concentration in the total mass of ingested media was calculated for each congener or homologue group for each receptor. The method for calculation is analogous to the approach used for calculation of exposure via food ingestion for fish (Section 4.3.1.1), shown in Equation 4-4. The congener or homologue concentration in each ingested medium was weighted according the fractional contribution of the media type to the total mass of ingested media (Table 3-12). Resulting congener concentrations or homologue concentrations were then used as input into the regression models (Table 4-14). Egg concentrations for each congener or homologue group were then estimated using the regression equations published by Elliott et al. (2001) (Table 4-14). Because calculations were conducted using laboratory-reported homologue concentrations, and not sums of individual congeners, resulting egg $TEQ_{DF,B}$ concentrations are expected to have an upward bias. The degree of bias is unknown due to variability in the results for individual samples, but is higher when all congeners in a homologue group were detected, and indeterminate when congeners and homologues were reported as nondetects. Uncertainties associated with use of the Elliott et al. (2001) regression models are discussed further in Section 7.

Concentrations of congeners in homologue groups for which Elliott et al. (2001) did not publish regression equations were estimated by using regression parameters for the most closely associated homologue group (e.g., HpCDF was modeled using the equation for HxCDF). This substitution allowed prediction of congeners or homologue concentrations in eggs for all congeners except the octachlorinated congeners. Octachlorinated congeners have rarely been reported in bird tissue (see Table 15 of Integral 2010b), and have very low TEFs (Table 3-2). Moreover, they are the largest among the dioxin and furan congeners and therefore the least bioaccumulative (Integral 2010b). For these reasons, the lack of predicted egg concentrations of octachlorinated congeners is expected to have a negligible effect on the final egg $TEQ_{DF,B}$ concentration estimates. Further, because regression parameters in Elliott et al.'s (2001) models for PeCDD and HxCDD are very similar (Table 4-14), the model substitutions that were made were considered appropriate. However, extending model substitutions for the octachlorinated congeners using one of Elliott et al.'s (2001) models was considered too uncertain because of the known differences from other congeners in the bioaccumulation patterns of the octachlorinated congeners.

Finally, estimated CT and RM concentrations of each congener or homologue in egg tissue were multiplied by the appropriate TEF to compute the final $TEQ_{DF,B}$ in eggs. Because two homologue groups include congeners with different TEFs (Table 3-2), a conservative estimate of egg $TEQ_{DF,B}$ was calculated assuming the maximum TEF for all congeners in the group (Table 4-16). To estimate a lower bound on the estimated egg $TEQ_{DF,B}$ concentrations, a second calculation was performed using the lowest TEF for the homologue group, which resulted in a change to TEF for ∑HxCDD and ∑PeCDF (Table 4-16).

$TEQ_{DF,B}$ concentrations in eggs were calculated for prey consumption only, as well as with the inclusion of incidental sediment ingestion. The role of incidental sediment ingestion was evaluated under both baseline and post-TCRA conditions. In the post-TCRA exposure evaluation, concentration of dioxins and furans in the foods of birds were not changed from the pre-TCRA (baseline) scenario. This model was used only to estimate egg concentrations in the cormorant, heron, and sandpiper. Concentrations of dioxins and furans in the foods of killdeer were estimated using soil concentrations (Section 4.3.1.5.2 and Appendix D), and were regarded as an insufficiently robust foundation for further modeling to estimate egg concentrations. No estimates of killdeer eggs were prepared.

In response to USEPA comments on the draft of this report, example calculations showing each step and each parameter used in each example were prepared, and are presented as Exhibit 2A and 2B. Also in response to comments (Appendix F), estimates for egg concentrations were added for background conditions for sandpipers and herons consuming prey and shoreline sediments. Results of the TEQ calculations using the regression models from Elliott et al. (2001) to estimate concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are provided in Table 4-17.

The original models developed by Elliott et al. (2001) were based on concentrations in prey of piscivorous birds. Application of models to predict egg tissue concentrations from a mixture of different media including both prey and sediment is associated with some uncertainty. This uncertainty is discussed in Section 7.

4.3.2.2 Estimating PCB Concentrations in Bird Eggs

Although there is a wealth of literature on the biomagnification of PCBs from dietary sources into bird tissues, there have been few studies documenting specific biomagnification relationships for dioxin-like PCBs from foods of birds to their eggs. No studies were found that provide regression models of egg tissue on fish tissue or other food concentrations for dioxin-like PCB congeners. As a result, prey-to-egg BMFs are used to estimate dioxin-like PCB congeners in bird eggs. Moreover, no one study provides BMF for all dioxin-like PCB congeners, and no set of studies provides data for the full suite of dioxin-like PCB congeners for any one bird species. Given the uncertainties that already result from the use of BMFs, combining BMFs for different species across different studies to generate a suite of BMFs for all dioxin-like PCB congeners was considered prohibitively uncertain. Instead, estimates of concentrations of only a subset of dioxin-like PCB congeners in bird eggs are developed and presented in this BERA. The result underestimates the role of PCBs in risks to birds, but a means to comprehensively address dioxin-like PCBs in bird eggs was not available. The degree of the underestimate is likely small, because the selected congeners are those with the highest TEFs.

4.3.2.2.1 Overview of Literature Found

On the basis of the available literature, prey and egg data or congener-specific BMFs were extracted from the literature to estimate concentrations of selected PCB congeners in bird eggs. Although several papers address the PCB congeners with the highest dioxin-like potency (PCB77, PCB81, and PCB126), two of these congeners were detected rarely in sediments collected from the Site (Table 4-18), and in some cases, were not detected at all outside of the original 1966 perimeter of the impoundments north of I-10. Six of the 12 dioxin-like PCB congeners were ultimately selected for modeling egg concentrations: the three with the highest TEFs regardless of detection frequency: PCB77, PCB81, and PCB126; and three with relatively high detection frequencies in Site sediments and relatively high TEFs: PCB105, PCB114, and PCB118. Of the six selected congeners, concentrations of four of them correlate with concentrations of TCDD and TCDF in Site sediments (Integral 2011c), so measures to address risks from dioxins will address these congeners. Those that do not correlate with TCDD and TCDF were rarely detected in sediments.

Three sets of BMFs were used in this evaluation, to reflect the three different bird receptor surrogates. BMFs for herring gull were taken from Braune and Norstrom (1989). Braune and Norstrom (1989) data include only a limited set of PCB congeners, and only two of the dioxin-like PCBs, PCB105 and PCB118. Results were considered analogous to eggs of the omnivorous cormorant (Table 4-18). Braune and Norstrom (1989) did not provide BMFs for all relevant congeners, but they provide data for three tetrachlorinated PCB congeners and five pentachlorinated PCB congeners. To estimate BMFs for the congeners selected, an average of BMFs within these homologue groups was calculated, and applied to the PCB congener within the same homologue group. This was necessary to estimate BMFs for PCB77 and PCB81 (tetrachlorinated PCBs) and for PCB114 (a pentachlorinated PCB) (Table 4-18).

Congener-specific BMFs for the gray heron (*Ardea cinerea*), a bird nearly as large as the great blue heron, and for the kingfisher (*Alcedo atthis*), a smaller piscivore, were compiled by Naito and Murata (2007) from Murata (2003) and Murata et al. (2003), and used to represent the great blue heron and sandpiper, respectively. Although important differences from the receptor surrogates are recognized, the results are considered to reflect general estimates of $TEQ_{P,B}$ for the various bird eggs, and to generally represent the variability in this parameter.

Selection of data for input to the BMF models was conducted in the same manner as selection of data for input into the regression models for dioxins and furans: An individual sample of each medium was selected to represent the CT and RM exposures. To do this, the CT and RM $TEQ_{P,B}$ concentrations of each medium within each exposure unit were calculated. The actual sample with the $TEQ_{P,B}$ concentration closest to the CT and the sample with the $TEQ_{P,B}$ concentration closest to the RM were identified. The specific congener concentrations within these samples were used in subsequent calculations.

Similarly, the prey-weighted average concentration of each PCB congener for the total mass ingested by each bird receptor was calculated, using the same approach used to compute the final input for the dioxin and furan egg model. Once a total ingested concentration of the PCB congener was calculated, it was multiplied by its respective BMF (Table 4-18) and the resulting TEQs were summed for a total $TEQ_{P,B}$ concentration. All TEF values are presented on a ng/kg ww basis, in Table 4-19. At the request of USEPA in its comments on the draft BERA, a series of examples of these calculations for each requested combination have been prepared and are presented in Exhibit 2A and 2B.

Given the manner in which the BMFs are derived, the range of studies used to provide parameter estimates, the variety of analytical methods and the general uncertainties associated with the use of BMFs, results of these calculations should be regarded as general estimates, useful only to provide perspective on the relative importance of PCBs in risks to birds across a range of bird species and feeding guilds. Results underestimate the $TEQ_{P,B}$ concentration in bird eggs because not all congeners could be modeled. Results provide a perspective on the relative importance of PCBs in TEQ risk to birds under the baseline condition, with and without the influence of sediment ingestion for cormorants, herons, and sandpipers as well as under post-TCRA and background conditions (cormorant only).

4.3.2.3 Egg Exposure Scenarios

A total of five different scenarios were modeled to determine baseline risks to birds from dioxins, furans, and PCBs. The details of each scenario are:

- **Prey:** Ingestion of prey is the only source of dioxins and furans or PCBs and follows ingestion parameters detailed in Section 4.3.1. Evaluation of exposure to dioxin-like chemicals via prey ingestion is useful only for determining the relative importance of sediment exposures.
- **Prey and sediment:** Results of this analysis represent the baseline exposure assessment for this line of evidence. The sediment ingestion rate for sandpipers is appreciable, with lesser sediment ingestion rates for great blue heron and cormorants (Table 3-12). In all birds, sediment ingestion will contribute to the overall intake of dioxins, furans, and PCBs. This scenario takes sediment ingestion into account and uses the same exposure assumptions and exposure units as for the wildlife exposure model (Section 4.3.1). Cormorants were assumed to ingest sediment from 0 to 6 inches from the aquatic and shoreline areas of the Site, excluding sediments from the western cell of the impoundments. All shoreline sediment (0 to 6 inches) samples for the site were included for the great blue heron and sandpiper.
- **Prey and sediment (post-TCRA):** For analysis of post-TCRA exposure, all samples for stations within the original 1966 impoundment perimeter are removed and replaced with a single CT or RM value of TEQ_{DF} or TEQ_{P} to represent the possible post-TCRA condition (Appendix C). Regression models were not available to estimate post-TCRA whole crab and whole catfish concentrations, so the baseline dataset for tissue

was used in models of the post-TCRA egg concentrations, resulting in a conservative assessment of post-TCRA exposure. Because there are no data for PCB congeners in shoreline sediments from upstream, post-TCRA estimates for $TEQ_{P,B}$ on the Site, which represents the post-TCRA sediment condition using median background concentrations, could not be made for the herons and sandpipers. Upstream benthic sediments were included in background calculations for cormorant.

- **Background (prey only):** For comparison with the Site, background exposures were modeled. Background analysis used all tissue data collected from the background areas. The CT and RM of TEQ_{DF} and TEQ_{P} from the background dataset were used for data selection and were determined independently from upstream data (Appendix C).
- **Background (prey and sediment):** Background egg concentrations were also estimated including ingestion of both prey and sediment. Tissue and sediment (0 to 6 inches) data from upstream background study were used. The reference TEQ_{DF} or TEQ_{P} used for data selection was calculated independently for background conditions (Appendix C). Because there are no data for PCB congeners in shoreline sediments from upstream, background estimates with exposure to PCBs from both sediment and prey could not be made for the herons and sandpipers. Upstream benthic sediments were included in background PCB calculations only for the cormorant.

The regression models used for the dioxin and furan estimates and the BMFs used for the PCB calculations were based on concentrations in prey of piscivorous birds. Modeling using regression equations or BMFs based on exposure via fish ingestion to predict egg tissue concentrations from exposure via a mixture of different media including both prey and sediment is associated with some uncertainty. This uncertainty is discussed in Section 7.

4.3.2.3.1 Data Used to Estimate Bird Egg Concentrations

Data used was identical to that used in the wildlife exposure model detailed in Section 4.3.1 with the exception that no soil data were included, all tissue data were used on a wet weight basis (sediment values were dry weight) and all scenarios were tested regardless of resultant HQ values.

4.3.2.3.2 Results of Bird Egg Models

Estimated $TEQ_{DF,B}$ concentrations in the eggs of the neotropic cormorant, great blue heron, and spotted sandpiper are shown in Table 4-17. Estimates of $TEQ_{P,B}$ concentrations in the eggs of the neotropic cormorant, great blue heron, and spotted sandpiper are shown in Table 4-19. The $TEQ_{DFP,B}$ for eggs of each receptor, showing the relative importance of PCBs and dioxins and furans for each scenario, are provided in Table 4-20.

4.4 Probabilistic Exposure Assessment

Probabilistic exposure assessment was performed for receptors whose estimated exposure for one or more $COPC_E$s equaled or exceeded the associated LOAEL in the deterministic risk assessment (i.e., spotted sandpiper, killdeer, and marsh rice rat). The probabilistic exposure assessment involved assigning probability distributions to certain exposure parameters to yield a probability distribution for $COPC_E$ exposure. This exposure distribution was then compared to the TRV, and the likelihood that exposure exceeded the TRV (under the assumptions used) was determined. Exposure distributions were developed for the site as a whole for each relevant receptor–$COPC_E$ pair.

Probabilistic analyses of exposure and risk were developed using Oracle® Crystal Ball software (Gentry et al. 2005). Crystal Ball employs Monte Carlo analysis, a commonly used probabilistic numerical technique where the uncertainty and variability in exposure (and HQ) estimates are characterized by estimating the exposure (and HQ) distributions. To develop each exposure distribution, the exposure estimate for a receptor–$COPC_E$ pair is repeatedly calculated by Crystal Ball, with each iteration of the exposure model using different sets of parameter values determined by random sampling of the probability distributions for those input parameters treated probabilistically (USEPA 2001). Those parameters modeled probabilistically and the means used to estimate the exposure probability distributions for applicable receptor–$COPC_E$ pairs are discussed further below.

4.4.1 Parameters to be Estimated in a Probabilistic Analysis

Certain receptor-specific exposure parameters identified on the basis of life histories, and Site-specific EPCs used in the wildlife exposure model were treated probabilistically to increase understanding of ecological risk. Parameters treated with probability distributions included EPCs, body weight, feeding rate, prey fraction for each prey type, water ingestion rate (when relevant), and rate of incidental ingestion of soil or sediment. Additional life history information from the literature was required to perform the probabilistic analyses, because parameter estimates for the probabilistic analyses require measures of variance and range. Results of the information search to obtain the required data are presented in Table 4-21.

Because EPCs associated with terrestrial prey for upland receptors were estimated using BAFs from Site soils, the contribution of prey in these cases is also dependent on the underlying Site soil data used to derive the prey component of the diet. Therefore, $COPC_{E}s$ in terrestrial invertebrate tissue were varied probabilistically with soils.

4.4.2 Derivation of Parameter Distributions

To derive parameters for distributions of Site-specific EPCs, relevant $COPC_{E}$ soil and sediment concentration data were compiled and imported into Crystal Ball for distribution goodness-of-fit testing. Goodness-of-fit testing employed Anderson-Darling, Chi square, and Kolmogorov-Smirnov analyses for ranking the fit of each $COPC_{E}$ dataset against 14 available distribution types. Distributions selected by Crystal Ball for each dataset were compared to distributions selected by other means (e.g., R [R Development Core Team 2008]), if available. If no other distribution information was available for a given dataset, only Crystal Ball was used to evaluate its distribution. If the distribution recommended by Crystal Ball for a particular dataset differed from the recommendations of other software programs, professional judgment was used to select the best fitting dataset distribution. Following selection of an appropriate distribution for each soil or sediment concentration dataset, distribution parameters were estimated by Crystal Ball and incorporated into the probabilistic model.

To derive parameter distributions for life history parameters, the CT and range of each parameter were determined from the literature. Where assignment of a normal distribution was appropriate (e.g., for body weight of a receptor), the mean and standard deviation were derived from the literature. For the prey fraction for each prey type and the rate of incidental ingestion of soil or sediment, a triangular or uniform distribution was assigned using the estimates for the CT and the range (only the range was needed in the case of the uniform distribution). For these exposure model terms, professional judgment was used to derive the range when data were not readily available in the literature. A normal distribution is defined by a mean and a variance (or standard deviation). A triangular distribution is defined by a mode, a minimum, and a maximum. A uniform distribution is defined by a minimum and a maximum. The triangular and uniform distributions are used when information is limited and the form of the distribution is unknown. For feeding rate and water ingestion rate, allometric equations were applied to determine the appropriate value corresponding to the body weight value randomly selected during a given iteration of the Monte Carlo.

Distribution characteristics used in probabilistic risk analysis are summarized in Appendix C for EPCs and in Table 4-21 for other exposure parameters.

5 EFFECTS CHARACTERIZATION

Lines of evidence in this BERA employ both TRVs, which are intended to denote no-effects/effects thresholds for survival, growth, and reproduction of individuals; and benchmarks such as the AWQC, considered protective of a broader group of taxa (e.g., aquatic macroinvertebrates or aquatic communities). Detailed information on the methods used and data considered in selection or derivation of NOAEL and LOAEL TRVs and benchmarks used in this BERA is provided in Appendix B. This section provides an overview of the types of TRVs and benchmarks used in calculating HQs, methods to aggregate toxicity data, approaches to selection of each TRV needed, the general meaning of different types of TRVs or benchmarks, and types of uncertainties common to these approaches.

5.1 Types of Toxicity Information Used

Selection of TRVs and benchmarks for use in ecological risk assessment involves consideration of several factors: types of receptors under evaluation and assessment endpoints for each, whether the analysis calls for a screening or a more realistic risk description, the data and methods available for estimating exposure to receptors, and the availability of toxicity information that meets basic data quality standards. To address all of the lines of evidence for each receptor to be used in this BERA, effects measures consisting of TRVs or benchmarks expressed in the following terms are needed:

- Bulk sediment concentrations (mg/kg) that are protective of the benthic macroinvertebrate community
- Concentrations in water (mg/L) that are protective of benthic macroinvertebrate communities and fish
- Concentrations of metals in media ingested by fish (mg/kg)
- CTR values for TCDD (or other organic compounds) as concentrations in whole fish (mg/kg ww)
- CTR values for TCDD (or other organics) expressed as concentration in whole clams (mg/kg ww)
- CTR values for dioxins, furans and PCBs expressed as a TCDD or TEQ concentration in eggs of birds (ng/kg ww)
- Daily ingested dose NOAELs and LOAELs (mg/kg bw-day) for reptiles, birds, and mammals for all COPCs.

When using published toxicity literature to establish measures of effect, the specific meaning of the effects measure depends on the experimental design used and the test endpoints. For example, a toxicity study may provide a threshold dose above which a reduction in the hatchability of bird eggs occurs, or a reduction in the growth of juveniles. Exceedance of TRVs from such studies would have different meanings to the risk assessment. In cases where the estimated exposure to an ecological receptor is greater than an LOAEL and risk to the receptor cannot be considered negligible, the specific endpoint represented by the TRV or benchmark is considered in the description of risk. A risk estimate based on a TRV denoting an LOAEL for effects on survival is interpreted to have a potentially more severe effect on the receptor than exceedance of a TRV denoting an LOAEL for individual growth rate or reproduction.

In some cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV was required (e.g., Table 5-1). In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk. Although uncertainty associated with estimating an NOAEC from an LC_{50} [median lethal concentration], which was required for this risk assessment in some cases, may be substantial, Chapman et al. (1999) do not support the use of uncertainty factors greater than 10. They also clearly avoid specific recommendations for uses of uncertainty factors, focusing instead on general technical considerations in their use, and point out that their use does not specifically resolve uncertainty, it can only compensate for a lack of empirical information. Chapman et al.'s (1999) discussion is summarized in Appendix B, and related uncertainties are addressed in Section 7.

5.2 Methods Used for Aggregation of Toxicity Data

As described in Appendix B, many TRVs used in this risk assessment were those presented in compendia of values prepared by federal agencies (e.g., Sample et al. 1996; T&N Associates 2002; USEPA 2005a) or from USEPA-approved, final risk assessments conducted for other CERCLA sites. In most cases, the final selected TRV (NOAEL or LOAEL) was either the geometric mean of data from studies of acceptable quality (e.g., the TRVs developed by USEPA and others for the ecological soil screening levels [EcoSSLs]), or in cases where insufficient information was available to calculate a geometric mean, the TRV was the lowest LOAEL and the highest NOAEL from among studies of acceptable quality. If the highest NOAEL was greater than the lowest LOAEL, then the highest NOAEL that did not exceed the lowest LOAEL was selected.

This approach results in fairly conservative estimates of toxicity and a fairly protective risk assessment overall. For dioxins and furans, and for PCBs because the toxicity of some congeners is considered to be additive with that of TCDD, the relatively extensive literature available was reviewed in greater detail. In both cases, more than one TRV of acceptable quality are often available for certain species. For example, there are several studies of PCB toxicity in mink, and there are several studies of TCDD toxicity to birds following injection into eggs. In cases such as these, if fewer than 10 values with a common endpoint and route of administration were found, the following steps were taken to derive a TRV, for example, a LOAEL:

1. Within-species LOAELs are grouped
2. The geometric means of the within-species LOAELs are calculated
3. Resulting geometric mean LOAEL values are pooled. No individual species is represented by more than one value, although some values are the results of only one study.
4. The geometric mean of the pool of data for multiple species is calculated, and that value becomes the LOAEL for the $COPC_E$ and receptor.

NOAELs were treated in the same way in cases of more than one acceptable study. This approach is consistent with calculation of TRVs for use in development of EcoSSLs (USEPA 2005a), and generally consistent with derivation of the benchmarks used. It results in values that are both representative of multiple taxa within broad categories of receptors, and reasonably conservative without being overly so.

The RI/FS Work Plan indicates that cumulative distribution functions derived from multiple effects-level metrics within a species, or SSDs, would be developed using multiple literature values for several species. This is a tool that can be used to clearly define the risk and the uncertainty associated with a risk calculation. However, sufficient data for a set of related taxa that have similar exposure and effects metrics were not found, except for the SSD for early life stage fish developed by Steevens et al. (2005). This SSD was the only one used in this BERA.

5.3 Benthic Macroinvertebrate Communities

For most $COPC_{ES}$, risks to benthic macroinvertebrate communities were estimated using benchmarks, either sediment quality guidelines (SQGs) or AWQC. SQGs expressed as bulk sediment concentrations for marine and estuarine environments were derived by Long et al. (1995) using data for a large number of contaminated sediment sites.

Although other sources of marine SQGs are available (MacDonald et al. 1996) and may be more robust on the basis of the methods used for their derivation, Long et al. (1995) is the same source of information used by TCEQ in establishing sediment screening benchmarks for benthos. TCEQ interprets sediment chemistry in terms of risk to benthic invertebrate communities relative to Long et al.'s (1995) sediment benchmarks as follows:

- The effects range-low (ER-L) values are concentrations below which adverse effects on benthic communities rarely occur
- The effects range-median (ER-M) values are concentrations above which adverse effects on benthic macroinvertebrate communities are "probable"
- At concentrations between the ER-L and ER-M, adverse effects on benthic invertebrates are considered possible.

Although Long et al.'s (1995) ER-L and ER-M values have technical flaws (e.g., Sampson et al. 1996a, 1996b; Becker and Ginn 2008), they are regarded by TCEQ as protective of benthic communities. Therefore, in this risk assessment and consistent with the role of SQGs as screening benchmarks, ER-Ls were used to identify $COPC_{ES}$ and stations posing negligible risk to benthic macroinvertebrate communities. When concentrations of a $COPC_{E}$ in sediment exceeds its respective ER-M value, the number of exceedances and area involved are considered to determine whether additional toxicity information is warranted to better describe risk.

AWQC are derived using a minimum dataset for at least 8 major aquatic taxa including invertebrates, fish, algae and vascular plants (USEPA 1985). AWQC are expressed as the criterion maximum concentration (CMC) for evaluation of short term (1-hour) spikes in chemical concentrations, and the criterion continuous concentration (CCC), a concentration that can be present for long periods with no adverse effects. These criteria are considered to be protective of 95 percent of aquatic species when concentrations of a chemical are not present in surface waters above the CMC and CCC within their specific time limits. The CCC was used for comparison to estimated concentrations in porewater for those chemicals lacking ER-L and ER-M values.

Neither type of benchmark is available for carbazole, phenol, BEHP, barium, cobalt, or manganese. For these chemicals, information searches were conducted as described by Appendix B, primarily using USEPA's ECOTOX database. The ECOTOX database includes results of studies that could be used to derive water quality criteria if all of the required taxa were represented. For the most part, available data included LC_{50} concentrations, and uncertainty factors were used to derive concentrations below which no effect was expected. Details are provided in Appendix B.

Attachment B2 to the RI/FS Work Plan provides a detailed summary of studies testing the toxicity of dioxins and furans to benthic macroinvertebrates, and that information is presented again in Appendix B to this document (summarized in Table B-4 of Appendix B). From this information, a no-observed-adverse-effects concentration (NOAEC) for TCDD in sediments was derived as the geometric mean of all NOAECs found in the literature for a wide range of invertebrate taxa.

In addition, a CTR for interpretation of reproductive risk to molluscs was found, and is included as the TRV for comparison to concentrations in clams. Wintermeyer and Cooper (2007) report that at 2 ng/kg in female eastern oysters (*Crassostrea virginica*), initial development of follicular structures and oocytes were notably different than in controls. Males had normal gametogenesis at this concentration. At 10 ng/kg ww, marked effects on gonad development and gamete maturation relative to controls were observed in both male and female oysters, as well as morphological lesions in females leading to resorption of oocytes. Effects were more evident at lower doses in females and were more pronounced in both males and females at 10 ng/kg than at 2 ng/kg. Cooper and Wintermyer (2009) report that in clams, the majority of TCDD is found in tissue of gonads 28 days after exposure, supporting the observation that TCDD affects these tissues. Other publications by these authors provide added detail. Wintermyer and Cooper (2003) present both a field study and a laboratory experiment with the eastern oyster. Although they report that 2 ng/kg is also associated with reduced veliger larval survival, this may overstate the effect of TCDD in the field study, because Wintermyer and Cooper (2003) used test subjects that were field-collected and field-exposed. Exposures related to their tests occurred in an urban estuary in New Jersey, but Wintermyer and Cooper (2003) did not document exposures to other environmental pollutants there, which could include PAH, estrogenic compounds, and physical stressors such as siltation. Therefore, the effects levels they report from their field study could overestimate the role of TCDD. However, Wintermyer and Cooper (2003) also exposed oysters to TCDD (without other chemicals) in a controlled experiment, and found reduced egg fertilization success, and reduced larval survival at the lower concentration in the adult tissue (2 ng/kg ww). Therefore, although the field study cannot account for the effects of the chemical mixtures, the laboratory study reported in this paper demonstrates that 2 ng/kg ww in whole eastern oyster tissue causes reduced fertilization of eggs and reduced larval survival in eastern oysters.

For this BERA, 2 ng/kg is considered the LOAEL for effects on reproduction in individual molluscs, as required by USEPA in comments (Appendix F). Because this tissue concentration is associated with a small but measureable histological effect that occurred only in females, reduced egg fertilization and reduced larval survival, this is a conservative TRV. A corresponding NOAEC was not available, and was not estimated. CTRs or other types of TRVs for other dioxin and furan congeners were not found.

More detailed information on the results of literature searches, derivation of TRVs, and all benchmarks is provided in Appendix B, and a summary of selected values is in Table 5-1.

5.4 Fish

The effects characterization for fish involved use of TRVs expressed as concentrations in foods of fish and in water For most metals, TRVs for interpreting concentrations in foods of fish were selected from literature reviews generated for ecological risk assessments approved for other CERCLA sites. Recent risk assessments for Portland Harbor in Portland, Oregon, and the Lower Duwamish Waterway in Seattle, Washington, provide extensive literature reviews. TRVs for BEHP and nickel expressed as concentrations in food were not available. Results of an acute toxicity test with sheepshead minnow were multiplied by an uncertainty factor to derive a no-effects concentration of BEHP for fish expressed as a concentration in water. For nickel, the results of tests with marine fish were combined to determine a chronic TRV for nickel expressed as a concentration in water. Details are provided in Appendix B. A summary of results of reviews to identify TRVs and benchmarks for fish is provided in Table 5-2.

To address the potential effects of dioxins and furans on fish, Integral used results of a study by Steevens et al. (2005). Steevens et al. (2005) developed an SSD to describe the toxicity of TCDD to several fish species, compiling multiple studies of TCDD and dioxin-like compounds with salmonids and other teleost fishes expressed as concentrations in fish eggs or embryos, a life stage that is sensitive to the effects of TCDD. Concentrations selected were those associated with no-observable effects, or the lowest concentration producing observable effects on egg survival. Steevens et al. (2005) selected the lowest paired effect levels available for a given species, calculated geometric means of the no-effect and lowest observable effect residue concentrations, and used the resulting 10 data points to derive the SSD. This risk assessment uses TEQ_F concentrations in whole body samples of fish for comparison to the CTRs of Steevens et al. (2005). This approach is conservative. Tietge et al. (1998) found that TCDD concentrations in eggs of brook trout (*Salvelinus fontinalis*) were 39 percent of the concentrations in the whole fish. Heiden et al. (2005) reported an even lower level of egg accumulation of TCDD relative to female whole bodies in zebrafish, with egg concentrations of just 5 percent of whole adults. This risk assessment is conservative because it assumes a 1 to 1 ratio of whole adult fish to egg concentrations. Additional details are provided in Appendix B, including data used by Steevens et al. (2005).

5.5 Reptiles

Integral conducted a literature review to identify toxicity information useful for evaluation of risk to reptiles; details of the search methods and resources used are provided in

Appendix B. The majority of available studies report chemical concentrations in field-collected specimens, and provide no means of interpreting exposure in terms of the potential for harmful effects. There are studies describing the concentrations of PCBs, dioxins, and furans in tissue of turtles in which the authors evaluate correlations of chemical concentrations with embryo deformities. However, the presence of other chemicals in the animals studied, including organochlorine pesticides, confounds interpretation, and TRVs for reptiles could not be derived.

Some risk assessments have addressed this data gap by assuming that birds are an appropriate model for reptiles, and that TRVs derived for birds can be used to interpret exposure to reptiles. A recent publication by Weir et al. (2010) examines this assumption by comparing results of controlled laboratory tests on birds and reptiles for chemicals for which representatives of both groups have been tested, which are mostly pesticides and ordnance compounds (explosives). Weir et al. (2010) find that reptiles were more sensitive than birds in 5 of 15 cases and less sensitive in 3 of 15. The rest of the comparisons (7 of 15) were inconclusive, or birds and reptiles were approximately equivalent.

For these reasons, the absence of reptile-specific toxicity studies for the $COPC_{ES}$ at this Site and the uncertainties about their sensitivities relative to other receptors, this document does not specify TRVs or benchmarks for interpreting estimated reptile exposures. Risks to reptiles are addressed qualitatively, by considering their estimated exposures relative to exposures by other receptors, and by considering the overall patterns in risk estimates observed for the other receptors.

5.6 Birds and Mammals

Lines of evidence used to evaluate risk to birds and mammals include comparison of estimated daily ingestion rates for individual $COPC_{ES}$ at the Site to TRVs expressed in the same terms. Comparison of estimated concentrations of $TEQ_{DF,B}$ and $TEQ_{P,B}$ in bird eggs is also used to evaluate risks to birds. The methods to identify measures of effect for both of these lines of evidence are detailed in Appendix B and summarized below. Results of the process to identify and select TRVs for birds are summarized in Table 5-3; TRVs for mammals are summarized in Table 5-4.

5.6.1 Daily Ingestion Rate

The primary literature available to interpret estimated daily ingestion rates of COPCEs at the Site is highly variable in terms of age, quality, numbers and types of species studied, depth and completeness. Moreover, many ecological risk assessments have previously been conducted at CERCLA sites and these tend to draw from the same sets of studies, although there are some differences in the data quality considered acceptable among sites. Finally, USEPA and related federal agencies have compiled toxicity data for use in risk assessment (e.g., Sample et al. 1996) and for development of EcoSSLs (USEPA 2005a). For all of the COPCEs except dioxins, furans, and PCBs, this BERA initially draws from compendia of literature prepared by USEPA or affiliates, including Sample et al. (1996) and USEPA (2005; 2012), and two recent BERAs accepted by USEPA for CERCLA sites in Portland, Oregon, and Seattle, Washington. These were considered a reasonable starting place for identification of wildlife TRVs. The general literature accessible through standard search tools like PubMed, Biosis, Google Scholar and others was also consulted when established TRVs were lacking. A detailed description of how these resources were used is provided in Appendix B.

Although the assessment endpoints for this BERA are expressed in terms of populations, the vast majority of literature including studies employed by prior risk assessments address endpoints on the level of the individual organism. The types of individual effects measures derived from the literature for this BERA were limited to those clearly relating to population-level effects, generally the survival, growth, and reproduction of tested individuals. Effects on reproduction are interpreted to include developmental effects, when it is clearly related to the reduced survival of young. Studies addressing unrelated endpoints (e.g., cellular or biochemical alterations or modified gene expression) were not used to establish TRVs for the BERA, because these effects cannot be related to population-level assessment endpoints.

5.6.2 Egg Concentrations

Use of egg-exposure based TRVs is the recommended approach to risk assessment for birds by both TN & Associates (2002) and USEPA (2003b). USEPA (2003b) provides a compilation of results of toxicity tests in which exposures as concentrations in eggs were documented, building on the detailed literature review conducted by TN & Associates (2002) for USEPA's Office of Research and Development. Both laboratory and field studies were compiled by USEPA (2003b). A paper was only selected for use in USEPA's (2003b) analysis if it included all of the following:

- Evaluation of more than one quantitative dose or exposure level. Studies evaluating only one dose or exposure level were considered to have too much uncertainty.
- One or more quantifiable toxicological endpoint.
- Appropriate statistical tests showing significant changes in response with changes in dose or exposure levels.
- Evaluation of the potential for co-contaminants to affect results (for field studies).

USEPA's (2003b) compilation of TRVs expressed as TCDD (or TEQ) concentrations in eggs includes NOAELs for developmental impairment from laboratory studies ranging from 66 ng TEQ/kg egg for the chicken to 50,000 ng TEQ/kg egg for several other bird species, including two gull species, the Graylag goose, and the goldeneye (a duck). Corresponding LOAELs range from 150 to 4,400 ng TEQ/kg egg. Integral did not use all of these studies for developing egg tissue TRVs, as discussed in Appendix B.

Because of the selection criteria used by USEPA (2003b), Integral used studies compiled by USEPA (2003b) as a starting point. Toxicity data selected for interpretation of estimated bird egg concentrations were taken only from controlled laboratory studies in which TCDD was injected into yolks during the earliest stages of embryo development. Because there is known to be substantial inter- and intraspecies variability in response to TCDD and other dioxin-like compounds, and because there is evidence that the existing TEFs for birds may not fully describe the relative toxicity of various dioxin-like compounds (e.g., Cohen-Barnhouse et al. 2011), egg toxicity studies with other dioxin-like compounds were not used. To do so would have introduced variability in the estimate of toxicity to bird eggs with unknown effects and uncertainties. Finally, for development of the final TRV, only studies performed using yolk injection were used, because TCDD transferred from hens to eggs occurs only in the yolks (Nosek et al. 1992a). Selected TRVs from egg yolk injection studies are summarized in Table 5-5. Data from studies in which TCDD is injected into the albumin or the air cell were compiled and are discussed, but were not incorporated into the final TRV for eggs. Details on those studies, and relevant field studies are discussed and presented in Appendix B.

6 RISK CHARACTERIZATION

Risk characterization combines the information developed in the exposure and effects characterizations to provide quantitative and qualitative descriptions of the likelihood that hazardous materials at the Site are causing adverse ecological effects under the baseline condition for the Site. According to USEPA (1997) guidance, risk characterization should also present information important to interpreting risks (USEPA 1997).

Each risk question represents an independent line of evidence that was applied to address risks to each receptor. All lines of evidence involve the evaluation of whether estimated exposures on the Site exceed an exposure level or concentration associated with effects (Table 3-11). Factors contributing to interpretation of the exceedance include the adverse effect(s) represented by the benchmark or TRV exceeded, and the type of threshold exceeded (i.e., LOAEL, NOAEL, EC_{10}), and the quality of the toxicity data used. This section presents the results of these basic comparisons, and at the request of USEPA in comments on the draft BERA (Appendix F), includes a section devoted to evaluation of risks to threatened and endangered species and a discussion of bioaccumulation. These discussions are followed by a section providing analysis of uncertainty. The final section of this document provides a summary statement of risk that incorporates all lines of evidence for a given receptor to address risk questions, and addresses qualitative and/or quantitative analysis of uncertainty for each receptor.

6.1 Overview of Risk Characterization

As described in Section 3.8.1, this BERA uses a tiered approach to the analysis and characterization of risks: an initial assessment of risk is performed deterministically for each receptor–$COPC_E$ pair. The initial assessment is a reasonable worst case evaluation, resulting in an HQ for each receptor–$COPC_E$ pair. For each receptor–$COPC_E$ pair, subsequent analyses depend on the value of the HQ_L, with one of the following possible outcomes (Figure 3-4):

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level lower than the NOAEL (i.e., $HQ_N < 1$) is characterized as negligible.
- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level between the NOAEL and LOAEL (i.e., $HQ_N > 1 > HQ_L$) is characterized as very low, depending on the toxicity data supporting the NOAEL and LOAEL values

- Risk to individuals of any receptor from any $COPC_E$ to which the receptor is exposed at a level higher than the LOAEL (i.e., $HQ_L > 1$) is considered to be present. Risk to the assessment endpoint, which may be a population or community, is evaluated and discussed further in the context of the data supporting the TRV.

An HQ_L equal to or greater than 1 is interpreted to indicate a need for further evaluation of risk to the receptor using refined methods (e.g., more realistic exposure assumptions or probabilistic analysis) and/or additional data, and is considered in context of the specific toxicity information used to derive the TRV. In this case, subsequent analyses include:

- A probabilistic exposure evaluation
- Evaluation of post-TCRA risk
- Consideration of background.

For avian receptor–COPC pairs that are surrogates for protected species, potential exposures evaluated according to the method described in Section 4.3.1.6 are discussed in Section 6.7.

Deterministic COPC and receptor-specific HQs were calculated for the initial evaluation of risk, as described in Section 3.8. Methods to perform the probabilistic exposure analysis are presented in Section 4.4, and results are provided below for those receptor–$COPC_E$ pairs for which the deterministic HQ analysis suggests a potentially unacceptable risk. Evaluation of population-level risks is addressed qualitatively, and incremental risk relative to background is evaluated when the HQ_L is equal to or greater than 1.

6.2 Risks to Benthic Macroinvertebrate Communities

A summary of results for each line of evidence to assess risk to benthic macroinvertebrate communities is provided in this section.

6.2.1 $COPC_E$s in Sediment Relative to Benchmarks and the TCDD NOAEC

$COPC_E$s that were evaluated by comparing their concentrations in individual sediment samples with SQGs include copper, lead, mercury, and zinc. Concentrations of TCDD were compared to the NOAEC for sediments. Results are summarized in Table 6-1, and below:

- Copper does not exceed the ER-M at any sampling station. Copper exceeded the ER-L at one station within the 1966 perimeter of the northern impoundments, and at two stations adjacent to the Southwest Shipyard property, south of I-10 and to the east of the southern peninsula (Figure 6-1).
- Lead does not exceed the ER-M at any sampling station. Lead exceeded the ER-L at two stations south of I-10 and to the east of the southern peninsula (Figure 6-2). One station exceeding the ER-L is adjacent to the Southwest Shipyard property; the other in a shoreline sample across the channel, on the east bank of the San Jacinto River.
- Mercury does not exceed the ER-L or ER-M in any location outside of the 1966 impoundment perimeter. Mercury concentrations exceed the ER-L in four locations, and exceed the ER-M in two locations within the impoundment perimeter (Figure 6-3).
- Zinc does not exceed the ER-M at any sampling station. Zinc exceeds the ER-L at one station within the 1966 perimeter of the northern impoundments, and at two stations adjacent to the Southwest Shipyard property, south of I-10 and to the east of the southern peninsula (Figure 6-4). Zinc exceedances of the ER-L occurred in the same samples as copper (Figure 6-1).
- The concentration of TCDD in two 0- to 6-inch sediment samples exceeded the no-effects level for sediments, at Stations SJB1 and SJC1, both within the footprint of the TCRA. The NOAEC was calculated as the geometric mean of no-effects concentrations from spiked-sediment bioassays with a range of invertebrate taxa including polychaetes, bivalves, insects, and molluscs with growth and mortality endpoints. Exceedances of NOAECs are not interpreted to indicate risk; Table B-4 in Appendix B shows no-effects concentrations ranging up to 25,000 ng/kg.

Results for this line of evidence indicate that risks to benthic macroinvertebrates from copper, lead, zinc, and TCDD are negligible because concentrations exceed no-effects levels in very few locations, and do not approach the effects threshold (ER-M).

Exceedance of the ER-M for mercury in two 0- to 6-inch sediment samples within the impoundment perimeter does not indicate a widespread risk to benthos from mercury. Exceedance of the ER-M is not predictive of effects, but is interpreted by TCEQ to suggest that adverse effects are probable. Given the very limited area within which mercury concentrations exceed the ER-M, risks to benthic macroinvertebrates due to mercury are considered very low to negligible.

6.2.2 Estimated Concentrations in Sediment Porewater Relative to TRVs

$COPC_{ES}$ that were evaluated by estimating porewater concentrations at individual sampling locations and comparing these to AWQC or available TRVs include BEHP, phenol, cobalt, manganese, and thallium. Results are summarized in Table 6-1, and below:

- Estimated sediment porewater concentrations of BEHP do not exceed the NOAEC for BEHP at any location (Figure 6-5).
- Estimated sediment porewater concentrations of phenol exceed the NOAEC for phenol at five locations (Figure 6-6). However, phenol was not detected in the sixteen of the eighteen 0- to 6-inch sediment samples shown in Figure 6-6; in the remaining two sediment samples, phenol concentrations are estimated (J- or UJ-qualified). Because porewater concentrations for organic compounds were estimated on the basis of OC-normalized concentrations in sediment, exceedances of the phenol NOAEC in porewater at all five locations is an artifact of the low OC content in these samples. Phenol was not detected in any of these five locations.
- Estimated sediment porewater concentrations of cobalt do not exceed the NOAEC for cobalt at any location (Figure 6-7).
- Estimated sediment porewater concentrations of manganese exceed the estimated no-effects concentration at 12 locations distributed around the Site (Figure 6-8). Three of those locations correspond to exceedances of ER-Ls for copper and zinc, including 1 within the 1966 impoundment perimeter, and two adjacent to and to the east of the Southwest Shipyards, and an additional location adjacent to the Shipyards. Other locations where this occurs are distributed randomly around the site.
- Estimated sediment porewater concentrations of thallium do not exceed the NOAEC for thallium at any location (Figure 6-9).

Results for this line of evidence indicate that risks to benthic macroinvertebrates from BEHP, cobalt, and thallium are negligible because concentrations do not exceed no-effects levels in any locations. Risks due to phenol are also negligible because phenol was generally not detected or could only be estimated in sediment. Whether manganese presents a risk to benthic invertebrates is uncertain, because the only TRV was a no-effects level; there is no corresponding effect level to enable interpretation of potential effects. Spatial correspondence of the relatively elevated manganese with concentrations of copper and zinc above ER-L values suggest that sediments in small areas of the impoundments north of I-10, and sediments adjacent to the Shipyards contain metals at concentrations elevated relative to very conservative screening levels, but below concentrations that indicate risk. Results for copper, magnesium, and zinc do not indicate unacceptable risk.

6.2.3 TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs

Composite clam samples were collected at five transects on the Site (Figure 4-1) and two transects upstream (Figure 4-2). Concentrations of TCDD in clams on the Site, where detected, ranged from 0.647 ng/kg ww (J-qualified) in Transect 6 to 17.6 ng/kg in a sample from Transect 3 (Table 6-2). The five clam samples from Transect 3, which is directly adjacent to the impoundments north of I-10, have the five highest concentrations of TCDD among all clam samples, ranging from 5.79 to 17.6 ng/kg. The next highest concentrations are at Transect 5, collected directly adjacent to the upland sand separation area to the west of the northern impoundments, where the maximum TCDD concentration in clam tissue was 2.43 ng/kg. TCDD concentrations in two of five clam samples collected from Transect 5 were greater than 2 ng/kg ww, the lower threshold of effects on reproduction in molluscs (Appendix B). Concentrations of TCDD in clam tissue are highest where sediment concentrations under the baseline condition are highest, consistent with the finding reported in the PSCR (Table 6-61; Integral and Anchor QEA 2012) that, for the tetrachlorinated congeners, concentrations in sediment correlate significantly and relatively strongly with those in clam tissue (i.e., tau-b values of 0.67 and 0.71 for TCDD and TCDF, respectively at $p < 0.05$).

The TRVs available to interpret tissue concentrations in molluscs are based on a series of studies in which oysters were injected with TCDD at various doses, and reproductive tissues were analyzed to determine if adverse effects on gametogenesis would result from TCDD exposure. In separate studies, clams were collected from the field and tissues observed, and were injected with radiolabeled TCDD to evaluate toxicity and bioaccumulation (Wintermeyer and Cooper 2003; Cooper and Wintermyer 2009). Wintermeyer and Cooper

(2007) report that at 2 ng/kg in female oysters, initial development of follicular structures and oocytes were notably different than controls. Males had normal gametogenesis at this concentration. At 10 ng/kg ww, effects were more pronounced, and were evident in both males and females. Female reproductive tissues were more sensitive to TCDD exposures; effects were more pronounced in both males and females at 10 ng/kg than at 2 ng/kg. Cooper and Wintermyer (2009) found that in clams, the majority of TCDD is found in tissue of gonads 28 days after exposure, supporting the observation that TCDD affects these tissues.

Cooper and Wintermyer (2009) also summarize other studies on this subject, including Wintermeyer and Cooper (2003), which involved field and laboratory components. In the field study, the authors transplanted adult eastern oysters to Newark Bay, the Arthur Kill area of Raritan Bay, and Sandy Hook, New Jersey. Results suggest that oysters with TCDD (ng/kg)/TCDF (ng/kg)/total PCB (µg/kg) concentrations of 3.2/2.1/68 and of 1.3/1.7/65 had reduced survival of veliger larvae. Conditions of this study are not analogous to conditions at the SJRWP because of the relatively high levels of PCBs in the oyster tissue, which could have been the cause of reductions in larval survival. Also, the field study reported by Wintermeyer and Cooper (2003) exposed test organisms in complex urban estuaries, where sediment and water quality are influenced by oil refineries, urban runoff, combined sewer overflows, sewage treatment plants, and other sources of anthropogenic pollutants. The effects of estrogenic compounds and other chemicals in addition to TCDD, TCDF, and PCBs were not considered or discussed by Wintermyer and Cooper (2003), and exposures of test organisms to other chemicals were not evaluated.

However, in the laboratory, Wintermyer and Cooper (2003) injected eastern oysters with TCDD, with resulting nominal tissue concentrations reported at 2 and 20 ng/kg ww. Oysters exhibited a dose-dependent reduction in egg fertilization success and in larval survival. Therefore, this paper demonstrates that 2 ng/kg ww in whole bivalves causes reproductive effects in addition to the histopathological effects observed in female oysters at this exposure level (Wintermyer and Cooper 2007).

All five clam samples collected adjacent to the northern impoundments at Transect 3 had tissue concentrations higher than 2 ng/kg, and four out of five at this location had tissue concentrations higher than 10 ng/kg. Two out of five (40 percent) clam samples next to the upland sand separation area were just above the 2 ng/kg, the LOAEL for histological effects in individual females and reduced egg fertilization and larval survival (Table 6-2). Although it is not possible to specify the effect on mollusc populations, individual clams from the area represented by Transect 3, assuming they are as sensitive as the oysters of Wintermyer and Cooper (2007), are at risk of reproductive impairment. Because of uncertainty associated with the use of literature-based TRVs, the field logbooks for collection of the clams were consulted. Field notes for the clam sampling for this project indicate no difficulty in capturing clams at Transect 3; clam collection required about 30 minutes at each of the transects (Integral 2011e), regardless of where they were collected. Although this is not the result of systematic study, any long term population level effects due to reproductive impairment in clams would suggest that capture of clams at Transect 3 should be more difficult, which was not the case. In light of this anecdotal information, although reproductive risk to individual clams collected from Transect 3 is present, risk to mollusc populations is considered low.

TCDD concentrations in three of five samples of clams collected adjacent to the upland sand separation area (Transect 5) exceed the reproductive LOAEL for oysters, but TCDD concentrations in 60 percent of clam samples from Transect 5 were below concentrations associated with effects on reproduction in individuals. The concentrations in the remaining 40 percent were just above the lower threshold of effects, indicating a substantially lower risk than at Transect 3. Therefore, risks to individual molluscs collected from Transect 5 appear to be low, and risks to populations are negligible. Risk to molluscs collected at Transects 2, 4, and 6 are negligible, because TCDD concentrations in clam tissues of these transects are below the LOAEL for the histological endpoint identified by Wintermeyer and Cooper (2007) and reproductive endpoints reported by Wintermyer and Cooper (2003).

It is not possible to evaluate post-TCRA risk to clams in the vicinity of Transect 3, but there is a statistically significant correlation between sediment TCDD and clam tissue TCDD for the site (Integral and Anchor QEA 2012). Because the concentrations of TCDD decline rapidly with distance from the impoundment, it is likely that the baseline risk of reproductive effects in individual molluscs is highly localized adjacent to the impoundments, and possibly only within the original 1966 impoundment perimeter. Because Transect 3 was within the TCRA footprint (Figure 4-1), it is also likely that risk to molluscs in the vicinity of

Transect 3 is greatly reduced as a result of the TCRA, which contained the area with the most contaminated sediments.

6.2.4 Miscellaneous $COPC_E$s

$COPC_E$s lacking a TRV or benchmark expressed as a concentration in porewater, tissue or sediment include carbazole and barium. $COPC_E$s for which reliable estimates in porewater could not be made are aluminum and vanadium. Results are summarized in Table 6-1, and below:

- Carbazole (Figure 6-10) in surface sediment samples from the site does not exceed concentrations upstream, although the maximum concentration upstream is estimated from the detection limit. Carbazole also was not detected in most locations on the Site and those concentrations that were detected are J-qualified (estimated).
- Aluminum (Figure 6-11) in surface sediment samples on the Site does not exceed the REV for aluminum except at one station, just beneath the I-10 bridge.
- Barium (Figure 6-12) in surface sediment samples on the Site exceeds the REV in 31 locations on the Site. The spatial pattern in sediments is random, with the highest concentrations from stations outside of the original 1966 impoundment perimeter. Barium in sediments does not appear to be associated with the impoundments north of I-10.
- Vanadium (Figure 6-13) in sediment samples on the Site exceeds the REV in 32 locations on the Site. The spatial pattern in sediments is random, and like barium, the highest concentrations are not within the impoundments north of I-10. Vanadium does not appear to be associated with the waste in the impoundments north of I-10.

Although specific toxicity information for carbazole is not available, the relatively small number of detects and small area with barely detectable concentrations suggests that carbazole does not present a risk to benthic invertebrates. Risks to benthic invertebrates on the Site from aluminum are not elevated over background. Although barium and vanadium are present in multiple locations on the Site at concentrations above the REV, the spatial distribution of samples with concentrations above the REV is random, and does not show an association with the impoundments. The highest concentrations of both barium and vanadium are outside of the impoundments, suggesting that the wastes in the impoundments are not the source of these metals. Any risks to benthic macroinvertebrates resulting from barium and vanadium are not associated with wastes.

6.2.5 Summary: Lines of Evidence for Benthic Macroinvertebrate Communities

Results of these analyses address the related risk question identified in Table 3-10: whether the concentrations of COPCEs in whole sediment from benthic habitats of the Site are greater than threshold concentrations relating to the survival, growth or reproduction of benthic invertebrates, or the productivity or viability of invertebrate populations or communities. Analysis results for benthic macroinvertebrates indicate that generally, they do not, although in localized areas adjacent to the former waste impoundments, tissue concentrations of TCDD in clams may affect reproduction of individuals. The area of impact, however, is small relative to the Site, so overall there is low risk to populations of molluscs, and only in a limited area, directly adjacent to the impoundments.

Risks to benthic macroinvertebrate communities from BEHP, phenol, copper, cobalt, lead, thallium and zinc are negligible. Risks due to carbazole and aluminum are no greater than in upstream areas. Risks to benthic invertebrate communities from barium, manganese, and vanadium, if any, have random spatial patterns not associated with the impoundments, and are therefore not a result of the presence of the impoundments.

Exceedance of the ER-M for mercury in two isolated surface sediment samples within the original impoundment perimeter does not indicate risks to the assessment endpoint for the overall benthic invertebrate community. Samples adjacent to affected samples are either below the ER-M or below the lower SQG, the ER-L. The isolation of these two samples, and the relatively small area affected, indicate negligible risk to benthic macroinvertebrate communities from mercury. In the post-TCRA environment, there are no risks to benthic invertebrates from mercury.

Risk to benthic macroinvertebrate communities from TCDD in sediments is negligible, according to the comparison of TCDD concentrations in surface sediments to the geometric mean of the NOAEC values (Appendix B). Since NOAECs available in the literature are a random assortment of values that are an artifact of the study designs of the publications from which they are drawn, an exceedance of the NOAEL is not considered to predict a potential effect. None of the studies of TCDD toxicity to invertebrates identified an effects concentration in sediment, even when 25,000 ng/kg was tested, so concentrations of TCDD in surface sediments from the Site cannot be compared to an effects level. Risks to the benthic community from TCDD overall is therefore considered negligible. Other dioxin and furan congeners cannot be evaluated because of a lack of toxicity data.

The analyses presented also address the following risk question (Table 3-10): whether concentrations of organic primary $COPC_{ES}$ (dioxins and furans) in tissue of field collected clams equal to or greater than concentrations considered threshold levels of reproductive effects in molluscs. Individual molluscs directly adjacent to the impoundment north of I-10 are at risk of reproductive effects from exposure to TCDD, and risks to populations at Transect 3 are considered low. Because tissue concentrations in all clams from this area (Transect 3) exceed the concentrations associated with effects in both male and female oysters, some effect on the reproductive productivity of clams or other molluscs in the area of very high concentrations of TCDD in sediment is possible. Although a precise estimate of the effect on the populations of molluscs on the Site is not possible, risks to molluscs from exposure to TCDD appear to be localized, and do not extend to other areas sampled elsewhere on the Site. Risks to a fraction of the individual molluscs near the upland sand separation area are very low, and risk to populations there are negligible. Risks to molluscs elsewhere on the Site are negligible. There are no toxicity data available to interpret tissue concentrations of the other dioxin and furan congeners.

Wintermyer and Cooper (2007) discuss possible mechanisms of the toxicity of TCDD to reproductive tissues of the oysters in their study, and acknowledge that the mechanism is AhR-independent. They are silent on the question of whether other congeners might have similar effects, but the absence of an AhR that binds dioxin in invertebrates indicates that the toxicity observed in oysters is not scalable to other congeners, as it is in birds, fish, and mammals. The potential effects of the other congeners are uncertain.

6.3 Risks to Fish

A summary of results for each line of evidence to address risk to fish is provided in this section.

6.3.1 Estimated Concentrations of Metals in Fish Diets Relative to TRVs

COPCEs evaluated by estimating the prey-weighted concentration in foods of fish (sediment ingestion was included) are those for which corresponding TRVs are available, and include cadmium, copper, mercury, and zinc. The analysis was conducted for the black drum and the southern flounder, expected to move around throughout the Site, and for Gulf killifish, on a smaller scale because these and related species are expected to have more localized foraging ranges. HQs for fish exposed to cadmium, copper, mercury, and zinc in foods and incidentally ingested sediment are summarized in Table 6-3. In no cases do the concentrations in ingested media exceed NOAELs or LOAELs for fish for these metals. Therefore, risks to all three fish receptors from cadmium, copper, mercury and zinc are negligible on the basis of this line of evidence.

6.3.2 Estimated Concentrations in Surface Water Relative to TRVs

COPCEs evaluated by estimating concentrations in surface water and comparing to TRVs for water are BEHP and nickel. Results are summarized in Table 6-4. An estimate of the Site-wide concentration of BEHP in water from a SWAC of surface sediments does not exceed the TRV for BEHP in water, which is a NOAEC. The estimated Site-wide concentration of nickel in water does not exceed the TRV for nickel, which was derived from several studies of marine fish (Appendix B). Therefore, risks to fish from BEHP and nickel are negligible on the basis of this line of evidence.

6.3.3 Total PCB Concentrations in Whole Fish Relative to the TRV for Fish

None of the whole hardhead catfish samples or Gulf killifish samples had total PCB concentrations above the NOAEC of 5.0 mg/kg ww or LOAEC of 16 mg/kg ww for total PCBs in fish (Table 4-5). Even the highest total PCB concentration, in a whole catfish from FCA 2, was more than a factor of 5 below the NOAEC. Risks to fish from total PCBs on the Site are negligible on the basis of this line of evidence.

6.3.4 TEQ Concentrations in Whole Fish Relative to the TEQ SSD for Fish

The analysis of toxicity data for fish eggs prepared by Steevens et al. (2005) and resulting SSD was used as the basis for comparison to Site-specific concentrations of $TEQ_{DF,F}$ and $TEQ_{DFP,F}$ (ng/kg lipid weight) in whole Gulf killifish and hardhead catfish. Results are considered representative of the fish receptors, the Gulf killifish, the black drum, and the southern flounder.

Representativeness of the hardhead catfish of the receptor fish species was evaluated by considering the available data for $TEQ_{DFP,F}$ in all fish tissues from the Site. Among all samples of fish from the Site (only the RI dataset includes whole fish, all other data are for fillet samples), which includes samples of the southern flounder, black drum, and several other fishes, hardhead catfish, gafftopsail catfish, and spotted sea trout dominate the upper end of the range of $TEQ_{DFP,F}$ concentrations (ng/kg ww) in fillet tissue. The relatively elevated concentrations in these species edible tissue samples (for which lipid data are not available for lipid normalization) could be caused by higher lipid content in edible tissue and not by greater exposures, but hardhead catfish are among the species with the highest $TEQ_{DFP,F}$ concentrations in fillet, suggesting that the hardhead catfish is a reasonably conservative representation of the southern flounder and black drum.

6.3.4.1 Killifish

There is no overlap in the distribution of concentrations of $TEQ_{DF,F}$ in whole killifish (Figure 6-14) with concentrations represented by Steevens et al.'s (2005) SSD. Therefore, there is no risk to Gulf killifish from dioxins and furans. When dioxin-like PCBs are included in the TEQ calculation, risks to Gulf killifish appear to be slightly increased (Figure 6-15; Table 4-6). One sample of whole killifish from Transect 4 has a concentration of $TEQ_{DFP,F}$ of 503 ng/kg lw, but this concentration is an artifact of high detection limits, and the true concentration is unknown. No dioxins and furans were detected in this sample, and PCB81, PCB123, and PCB169 were all not detected. If the estimated concentrations of these PCB congeners are removed, the $TEQ_{P,F}$ is only slightly reduced, to 193 from 196 ng/kg lw. The $TEQ_{DFP,F}$ concentration is below the concentration considered by Steevens et al. (2005) to be protective of 90 percent of fish species, but this comparison overstates risk, because none of the dioxins and furans in this sample were detected.

Therefore, there are negligible risks to Gulf killifish and those species that it represents resulting from dioxin and furan exposures alone, and there are generally negligible risks to these fish from the combination of all dioxin-like compounds. There is a small chance that the presence of dioxin-like compounds in one of two samples at Transect 4 could result in early life stage effects in killifish, but the available result for this sample is confounded by high detection limits, and a suggestion of risk is most likely an analytical artifact, because the other fish from this transect has a $TEQ_{DFP,F}$ concentration of 9.07 ng/kg lw, well below any risk threshold (Table 4-6). However, the $TEQ_{P,F}$ concentration of approximately 196 ng/kg lw is relatively high for this parameter. Transect 4 is near an outfall, which may affect the exposure of fish to PCBs in that area. Risk to Gulf killifish collected near the impoundments (Transect 3) is negligible.

6.3.4.2 Hardhead Catfish

There is no overlap in the distribution of concentrations of $TEQ_{DF,F}$ (ng/kg lw) on the Site with concentrations represented by Steevens et al.'s (2005) SSD for fish (Figure 6-16). Two samples of whole catfish from FCA 1 and one from FCA 3 have $TEQ_{DF,F}$ concentrations that slightly exceed Steevens et al.'s (2005) best estimate of the concentration at which 95 percent of fish species are protected (Table 4-6), and all samples are within the range of error of that calculation, suggesting a low to negligible risks to large fish represented by hardhead catfish from dioxins and furans. The result does not change appreciably when dioxin-like PCBs are added to the exposure estimate, except that $TEQ_{DFP,F}$ in two samples from FCA 2 also are equal to or slightly exceed the concentration protective of 95 percent of fish species (Table 4-6; Figure 6-17).

Given the conservatism of the Steevens et al. (2005) SSD for TCDD (because it is largely based on salmonids, which are known to be relatively sensitive to this and other toxicants), the conservatism of the approach, which assumes a 1 to 1 ratio of dioxin, furan, and PCB concentrations in whole fish to those of egg tissue (Tietge et al. 1995; Heiden et al. 2008), and that $TEQ_{DFP,F}$ in all samples is within the range of error of Steevens et al.'s estimate of the level protective of 95 percent of all fish species, risks to fish from exposure to dioxin-like compounds is very low to negligible.

6.3.5 Summary: Lines of Evidence for Fish

Risk questions for fish (Table 3-10) address whether the concentrations of $COPC_{E}s$ in waters of the Site, concentrations of inorganic $COPC_{E}s$ in the diet of fish, or concentrations of organic $COPC_{E}s$ in fish tissue from the Site are greater than the concentrations of $COPC_{E}s$ associated with the survival, growth or reproduction of fish. Analyses presented in this section indicate that they are not. Risks to all of the fish receptors from exposures to cadmium, copper, mercury, and zinc in the diet, including incidentally ingested sediment, are negligible. Risks to fish following exposure through water to BEHP and nickel are negligible. Risks to fish as indicated by total PCB concentrations in whole body samples are negligible.

Concentrations of $TEQ_{DF,F}$ (ng/kg lw) and $TEQ_{DFP,F}$ (ng/kg lw) in both whole Gulf killifish and whole catfish are generally below concentrations associated with adverse effects on fish early life stages. One Gulf killifish sample seems to exceed risk thresholds, but this is an artifact of elevated detection limits for dioxin and furan congeners. For five whole catfish, the $TEQ_{DFP,F}$ is slightly above the concentration protective of 95 percent of fish species, but within the margin of error, and below the concentration protective of 90 percent of species. Because the SSD derived by Steevens et al. (2005) is largely biased towards salmonids which are known to be among the most sensitive fish taxa for many toxicants, this evaluation is considered conservative. Overall, risks to fish on the Site are negligible.

6.4 Risks to Birds

Risks to birds were evaluated by comparing estimated daily ingestion rates of each $COPC_{E}$ to their respective TRVs expressed in the same terms. Risks to birds from exposures to dioxin-like compounds were also evaluated by comparing estimated egg concentrations to TRVs expressed as concentrations in eggs, providing a second and independent line of evidence to evaluate risks to birds from exposure to dioxins, furans and dioxin-like PCBs.

6.4.1 Estimated Daily Ingestion Rates Relative to TRVs

Results of the comparison of estimated daily ingestion rates of each $COPC_E$ by each avian receptor to its respective TRV are summarized in Table 6-5. For great blue heron and neotropic cormorant, daily ingestion rates of all $COPC_Es$ do not exceed NOAELs nor do they exceed LOAELs. This line of evidence indicates that risks to great blue heron, neotropic cormorant, and the species they represent, from ingestion exposure to cadmium, copper, mercury, nickel, zinc, BEHP, total PCBs, $TEQ_{DF,B}$, and $TEQ_{DFP,B}$ are negligible.

Estimated daily ingestion rates of cadmium, mercury, nickel, zinc, BEHP, and total PCBs by the spotted sandpiper also indicate that risks to sandpipers and the species they represent from these $COPC_Es$ are negligible. Estimated daily ingestion rates of copper by the spotted sandpiper could exceed the NOAEL, but neither the CT nor the RM exposures to copper for this receptor exceed the LOAEL. The avian TRV for copper was taken from the literature compilation in USEPA's EcoSSL for copper (USEPA 2007d), which identified over 3,000 papers and generated 393 copper TRVs for birds for a range of endpoints. The selected NOAEL of 4.05 mg/kg-day was the highest bounded NOAEL that was also lower than the lowest bounded LOAEL. The associated LOAEL from the study reporting the NOAEL of 4.05 mg/kg-day was 12.1 mg/kg-day for reproduction in chickens. Among the dataset compiled by USEPA (2007d), this NOAEL is among the lowest overall, and dozens of survival, growth, and reproduction NOAELs that are both higher than this and bounded by LOAELs are reported for sensitive endpoints in chickens as well as other species. The selected NOAEL for this risk assessment is from a study in which chickens were administered copper in food for 84 days and those exposed at the LOAEL exhibited a reduction in fecundity. Therefore, the selected TRV was a highly conservative representation of copper toxicity in individual birds, and exceedance of the NOAEL by a factor of 2 does not indicate a risk to sandpiper populations. Risks to this and other avian receptor populations from ingestion of copper are negligible.

The CT and RM of estimated daily ingestion rates of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ by the sandpiper exceed both the NOAEL and the LOAEL (Table 6-5). The HQ_L of 1 for CT exposure and the HQ_L of 3 for RM exposure indicates that there is a possibility that exposure of a shorebird foraging on the Site to dioxin-like compounds will be at levels that exceed effects levels for these chemicals. The very low HQs for $TEQ_{P,B}$ indicate that the risk to sandpipers is driven primarily by dioxins and furans, and not PCBs. Risks to sandpiper due to ingestion of dioxin-like compounds are evaluated further below.

Estimated daily ingestion rates of cadmium, copper, nickel, BEHP, and total PCBs by the killdeer also indicate that risks to killdeer and the species they represent from these $COPC_{ES}$ are negligible. Estimated daily ingestion rates of mercury exceed the NOAEL, but not the LOAEL. The study supporting the NOAEL (Heinz 1979) found no reproductive effects in the first generation of mallard ducks administered methylmercury dicyandiamide in the diet. Reproductive endpoints evaluated included fecundity and duckling survival. The study supporting the LOAEL used Japanese quail and reported reproductive effects at 0.9 mg/kg-day. Heinz (1979) administered methylmercury, which is highly bioavailable and is the toxic form of mercury, in the diet. In the killdeer exposure model, more than half of the daily mercury dose is derived from soil ingestion. However, methylmercury, the more toxic form of mercury, is generally not a large proportion of total mercury in soils, and thus, the Heinz study is not a realistic model of environmental conditions. Therefore, exceedance of the NOAEL by a factor of 2 (Table 6-5) does not indicate reproductive risk to individual killdeer. Risk to killdeer populations from mercury is negligible.

The RM ingestion rates of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ by killdeer are about equal to the LOAEL, indicating that risk to individual killdeer reproduction from dioxin-like compounds is present. The RM of the daily ingestion rate of zinc is about equal to the LOAEL for zinc in birds. The HQ_L of 1 for killdeer exposed to zinc indicates that there is a low probability that exposure of an individual terrestrial invertivorous bird foraging on the Site (prior to implementation of the TCRA) could occur at the effects level for zinc. Additional evaluation to describe risks to killdeer from zinc and dioxin-like compounds, including an evaluation of the probability that zinc and dioxin-like compounds exposures will exceed the LOAEL, is provided below.

6.4.2 Estimated TEQ Concentrations in Bird Eggs Relative to TRVs

Results of the evaluation of TEQ concentrations in the eggs of neotropic cormorant, great blue heron and spotted sandpiper relative to TRVs for egg mortality are summarized as HQs in Table 6-6, for all of the exposure scenarios modeled (Section 4.3.2). Concentrations of

TEQ in eggs of killdeer were not estimated because empirical data on the concentrations of PCBs, dioxins, and furans in their foods are not available. Results of risk calculations using this line of evidence are largely consistent with the results of risk calculations using estimated ingestion rates. Estimated concentrations of $TEQ_{DF,B}$ and $TEQ_{DFP,B}$ in the eggs of neotropic cormorant and great blue heron do not exceed the LOAEL concentration for egg mortality. Estimated concentrations in the eggs of cormorant do not exceed the field- or laboratory-based NOAELs for cormorants, except for the RM exposure that includes pre-TCRA sediment ingestion (Table 6-6, Tables 5-1 and 5-2). Similarly, estimated concentrations of $TEQ_{DFP,B}$ in eggs of great blue heron only exceed the NOAEL when ingestion of sediment is considered (Table 4-20). HQ_N values for great blue heron and cormorant ingesting prey and sediment are 2 to 3, but the egg-based HQ_L values for these scenarios are below 1 (Table 6-6). Results of several technically sound studies were used in deriving the egg TRV (Tables B-6 and B-8 of Appendix B). All of them report egg mortality as the endpoint. The final NOAEL for bird eggs was less than half of the lower of the two NOAELs available for cormorants, and the lowest effects level for cormorants was almost 10 times higher than the NOAEL (Table B-6). Therefore, an HQ_N of 2 for cormorants does not indicate risk of egg mortality in individual cormorants, and risk to cormorants is negligible.

There were no species-specific LOAELs for great blue heron, but a NOAEL of 207 ng/kg ww in eggs was reported for this species (Appendix B, Table B-9). The robust studies evaluating TCDD or TEQ in bird egg yolks report concentrations associated with actual effects that are from 2.2 to 12 times greater than NOAEL of 450 ng/kg. Also, there is substantial interspecies variability in the sensitivity to dioxin toxicity, and the relative sensitivity of herons is unknown. As a result, the HQ_N of 2 (or 3 at the RM exposure) is not a definitive indicator of risk or lack of risk to the mortality of eggs laid by individual birds. However, given the very conservative assumption that herons forage exclusively within its exposure unit on the Site, the inherent spatial bias of the associated sediment data set, and the conservatism of the egg model (Section 4.3.2.1.2), the egg exposure estimate is probably higher than the actual egg exposure. This is a key consideration given the uncertainty in the actual effects threshold for herons and that the exposure estimate is between the NOAEL and LOAEL. In light of the conservative representation of exposure, the egg-based HQ_N values for great blue heron are not interpreted to specifically indicate risk of egg mortality to individual herons.

The estimated post-TCRA egg concentrations for these two receptors indicate that implementation of the TCRA has a substantial effect on the potential exposures of these types of birds, reducing estimated egg concentrations. Baseline risks to neotropic cormorant and great blue heron from exposure to dioxin-like compounds is negligible.

The HQ_L values calculated using the baseline (prey plus sediment) CT and RM egg exposures for spotted sandpiper are consistent with results of those based on ingestion exposure: the CT and RM HQ_L values for this receptor are 1 and 2, respectively. This result indicates that the average egg exposure to shorebirds whose foraging habits result in extensive contact with sediments could equal the concentrations resulting in egg mortality, and the upper bound on the average egg concentration could be two times the LOAEL for egg mortality. The TRV used in the HQ_L calculation is the geometric mean of two other geometric mean LOAEL egg concentrations indicating egg mortality, one for ring-necked pheasants (1,215 ng/kg ww) and the other for double-crested cormorants (4,648 ng/kg ww) (Table 5-1). Egg mortality in these four studies ranged from 10 percent to 50 percent above that of controls. The results of a field study with spotted sandpipers indicated a NOAEL for egg mortality of 732 ng/kg ww (Appendix B), which is higher than the NOAEL used as a TRV, and higher than the NOAEL for pheasants, suggesting that the spotted sandpiper is not among the bird species considered highly sensitive to dioxin-like egg toxicity.

Results of both lines of evidence (estimated ingestion rate and estimated egg concentrations) are consistent in indicating some risk of egg mortality to the spotted sandpiper and the birds it represents from exposures to dioxin-like PCBs, dioxins, and furans. Risks to spotted sandpiper are considered in greater detail below.

6.4.3 Probability that Exposure Exceeds Effects Thresholds

A probabilistic analysis of exposure was conducted for those receptor–$COPC_E$ pairs for which the HQ_L is greater than or equal to 1. Probabilistic exposure analyses were conducted using only the wildlife exposure model, and not the egg exposure model. The exposure scenarios modeled probabilistically include zinc for killdeer, and $TEQ_{DFP,B}$ for spotted sandpiper.

6.4.3.1 Killdeer

A probabilistic exposure model for killdeer was performed for zinc and $TEQ_{DF,B}$ using the methods described in Section 4.4. Each of the resulting exposure levels generated by the Monte Carlo analysis was divided by the LOAEL for zinc. Results are presented as the cumulative probability distribution of the HQ_L for killdeer.

The result of the probabilistic exposure analysis indicates that there is an 8.3 percent probability that baseline exposures of killdeer to zinc will exceed the LOAEL (Figure 6-18).

The result of the probabilistic exposure analysis indicates that there is a 4.7 percent probability that baseline exposure of individual killdeer and the birds it represents to $TEQ_{DF,B}$ will exceed the LOAEL (Figure 6-19).

6.4.3.2 Spotted Sandpiper

Probabilistic exposure models for the spotted sandpiper were performed for ingestion of $TEQ_{DFP,B}$. The result of this analysis indicates that there is a 13.7 percent probability that baseline exposure of spotted sandpiper and the birds it represents to $TEQ_{DF,B}$ will exceed the LOAEL for wasting syndrome in adults and mortality of their eggs (Figure 6-20).

6.4.4 Post-TCRA Risks to Killdeer and Spotted Sandpiper

Under baseline conditions, zinc HQ_L values for killdeer equal 1, and TEQ HQ_L values for killdeer and spotted sandpiper exceed 1. These HQ_Ls were also calculated under post-TCRA conditions to determine whether implementation of the TCRA affects risk, and if the post-TCRA environment no longer presents risks to these receptors.

Table 6-7 provides a summary of pre-TCRA and post-TCRA HQ_Ls for these receptor–$COPC_E$ pairs. Risks to spotted sandpiper from exposures to $TEQ_{DF,B}$ using the line of evidence based on ingested dose are negligible in the post-TCRA scenario. The line of evidence based on estimated TEQ concentrations in eggs is consistent with the HQ_L results (Table 6-6). Therefore, implementation of the TCRA has eliminated risks to spotted sandpipers from exposure to $TEQ_{DF,B}$.

As for great blue heron baseline risks, the HQ_N for killdeer in the post-TCRA exceeds 1, but the HQ_L does not (Table 6-7). Similarly, although the absence of a species-specific threshold of effects for egg mortality results in some uncertainty, the several layers of conservatism in the exposure model for killdeer suggest that risks to individual killdeer from exposures to $TEQ_{DF,B}$ using the line of evidence based on ingested dose are very low in the post-TCRA scenario. Therefore, risk to the assessment endpoint, bird populations, is negligible. Risks to killdeer from exposure to zinc are not affected by implementation of the TCRA. This suggests that sources other than the waste impoundments are the primary source of this metal resulting in exposure to killdeer. Spatial patterns in surface soil concentrations of zinc within the exposure unit for killdeer support this conclusion: the samples with highest concentrations occur outside of the northern impoundments (Figure 6-21).

6.4.5 Risks to Killdeer and Sandpiper in Background Areas

The zinc and $TEQ_{DF,B}$ HQ_L values for killdeer equal 1, and $TEQ_{DF,B}$ HQ_L values for spotted sandpiper exceed 1 under baseline conditions. Risks in background areas are presented to provide perspective on the incremental risk to these receptors due to the Site. For the killdeer, the zinc HQ_L at the CT and RM background exposures are 87 and 71 percent, respectively, of the corresponding HQ_L values for the Site. This indicates that the incremental increase in exposure of killdeer to zinc at the Site is small, ranging from only about 13 to 29 percent, and suggests a substantial role of background conditions in the exposures of killdeer to zinc. The $TEQ_{DF,B}$ HQ_L at the CT and RM background exposures are 23 and 22 percent, respectively, of the corresponding HQ_L values for the Site, indicating that the incremental exposure in background areas is nearly a quarter of the exposure of killdeer to dioxin-like compounds on the Site.

For the spotted sandpiper, the $TEQ_{DF,B}$, $TEQ_{P,B}$ and $TEQ_{DFP,B}$ HQ_L values for background are low, regardless of whether the ingestion rate or the egg concentrations are considered (Tables 6-8 and 6-6). For both $TEQ_{DF,B}$ (and $TEQ_{DFP,B}$) baseline HQ_L values for background are about 1 percent of those on the Site, indicating that baseline (pre-TCRA) exposures of spotted sandpiper to dioxins and furans on the Site are substantially elevated over background. If background PCBs are considered on their own (as $TEQ_{P,B}$), the background $TEQ_{P,B}$ HQ_L values for CT and RM exposures are 26 and 21 percent, respectively, of those on the Site. This suggests that, although sandpipers (and related birds) are exposed to dioxin-like PCBs on the Site at levels higher than background, PCB exposures are a more important contributor to overall exposures of sandpipers to all of the dioxin-like compounds in background areas than they are on the Site.

6.4.6 Summary: Lines of Evidence for Birds

The analysis presented in this section addresses two risk questions (Table 3-10): 1) whether the total daily ingested dose (mg/kg-day) of $COPC_{ES}$ is greater than doses known to cause effects on the survival, growth, or reproduction in birds; and 2) whether the estimated concentration of dioxins and furans, expressed as TEQ_B, in bird eggs is greater than threshold concentrations for reproductive effects in birds. Results presented in this section indicate that there is a low probability that ingestion rates of zinc by killdeer, and ingestion rates of $TEQ_{DF,B}$ by the spotted sandpiper will exceed ingestion rates associated with adverse effects on bird reproduction. Results also indicate that $TEQ_{DFP,B}$ concentrations in eggs of sandpiper could also exceed those resulting in egg mortality. Ingestion rates of these and other chemicals by other bird receptors and estimated egg concentrations in the great blue heron and neotropic cormorant do not exceed effects thresholds.

Overall, baseline risks to individual birds on the Site are very low to negligible for most chemicals, and are low for dioxins and dioxin-like compounds. Baseline risks to cormorant and great blue heron are negligible for all of the $COPC_{ES}$, including dioxins, furans and dioxin-like PCBs, although there is some uncertainty about risks to heron due to a lack of species-specific effects thresholds for TEQ in eggs. Baseline risks to killdeer are negligible for all chemicals except zinc and dioxins and furans for which they are very low, and not much greater than background for zinc. Baseline risks to spotted sandpiper are negligible for all metals, BEHP, and total PCB as well as $TEQ_{P,B}$.

The probability that exposures of killdeer to zinc will exceed the effects level is low (8.3 percent). Background exposures to zinc are a substantial fraction of the overall exposure of killdeer to zinc. The probability that exposures of killdeer to dioxins and furans will exceed the ingestion-based LOAEL is low (4.7 percent). Background exposures to dioxins and furans represent about a quarter of the overall exposure of killdeer to dioxins and furans.

There is a moderate risk to spotted sandpiper from dioxins and furans under the baseline condition, as indicated by two independent lines of evidence, a wildlife exposure model of ingestion rate, and a model of egg concentrations. On the basis of a probabilistic evaluation of ingestion exposure, the probability that spotted sandpiper will be exposed to $TEQ_{DF,B}$ at levels exceeding TRVs is a moderate 13.7 percent. Although dioxin-like PCBs are additive with the $TEQ_{DF,B}$, the contribution of $TEQ_{P,B}$ to the exposure of spotted sandpiper is small, as indicated by both the estimated ingestion rate and the estimated egg concentration. Risks to the spotted sandpiper were reduced to negligible as a result of implementation of the TCRA.

6.5 Risks to Mammals

Risks to mammals were evaluated by comparing estimated daily ingestion rates of each COPC to their respective TRVs expressed in the same terms. Results are discussed below.

6.5.1 Estimated Daily Ingestion Rates Relative to TRVs

Results of comparisons of estimated daily ingestion rates of the $COPC_{ES}$ to their respective TRVs for the raccoon and the marsh rice rat are summarized in Table 6-9. Estimated daily ingestion rates of all $COPC_{ES}$ by raccoon are below LOAELs, regardless of whether the CT or RM is considered. Therefore, risks to raccoon, and the terrestrial mammals that it represents, are negligible.

Estimated daily ingestion rates of cadmium, copper, nickel, zinc, BEHP, and $TEQ_{P,M}$ and total PCBs by the marsh rice rat are all below their respective NOAELs and LOAELs, indicating negligible risk to the marsh rice rat for these $COPC_{ES}$. Estimated daily ingestion rates of mercury exceed the NOAEL but not the LOAEL (Table 6-9). The $TEQ_{DF,M}$ and $TEQ_{DFP,M}$ HQ_L values are both 2, indicating that marsh rice rats could be exposed to dioxins and furans at levels exceeding those resulting in reduced pup survival and effects on other reproductive endpoints in laboratory rats.

6.5.2 Probability that Exposure Exceeds Effects Thresholds

A probabilistic analysis of exposure of marsh rice rat to $TEQ_{DF,M}$ was conducted using the methods described in Section 4.4, and results are illustrated in Figure 6-22. There is a

14.3 percent probability that exposure of marsh rice rat to $TEQ_{DF,M}$ will exceed the level associated with effects on reproduction in mammals.

6.5.3 Post-TCRA Risks to Marsh Rice Rat

Exposures to the marsh rice rat following implementation of the TCRA is reduced to levels below those associated with effects, and the resulting $TEQ_{DF,M}$ and $TEQ_{DFP,M}$ HQ_L values are below 1 (Table 6-7), although NOAELs are still exceeded. The post-TCRA analysis conservatively assumes that concentrations in foods of rice rats do not change as a result of the TCRA (because post-TCRA food concentrations were not available). The reduction of the HQ_L to a value below 1 indicates that the majority of the rice rat exposures to dioxins and furans were associated with exposure to sediments within the impoundments.

6.5.4 Risks to Marsh Rice Rat in Background Areas

Exposure of marsh rice rat in background areas to $TEQ_{DF,M}$ as indicated by the HQ_L for background is very low (Table 6-8). The CT and RM exposures of marsh rice rat to $TEQ_{DF,M}$ in background areas are about 3 percent and 1 percent, respectively, of the CT and RM exposures on the Site, indicating that the incremental exposure of marsh rice rat to dioxins and furans at the Site is about the same as it is for the spotted sandpiper. Also like the sandpiper, the CT and RM exposures to $TEQ_{P,M}$ by the rice rat in background areas are about 37 and 29 percent of those on the Site, indicating that exposure of marsh rice rat to dioxin-like PCBs plays a larger role to the entire $TEQ_{DFP,M}$ exposure in background areas than it does on the Site.

6.5.5 Summary: Lines of Evidence for Mammals

Analyses presented in this section address the following risk question whether the total daily ingested doses (mg/kg-day) of $COPC_{ES}$ are greater than doses known to cause effects on the survival, growth, and reproduction of mammals. Results of the exposure and risk analyses indicate that, for all $COPC_{ES}$ except $TEQ_{DF,M}$, they are not, and that rates of ingestion of $TEQ_{DF,M}$ by raccoon do not exceed effects thresholds. Risks to raccoon are negligible for all $COPC_{ES}$. Risks to marsh rice rat are negligible for all $COPC_{ES}$ except TEQ. Risks due to $TEQ_{P,M}$ only are negligible, and dioxin-like PCBs do not contribute substantially to the $TEQ_{DFP,M}$ exposures. Marsh rice rats on the Site are at risk of reproductive effects and reduced survival of pups as a result of exposure to $TEQ_{DF,M}$. The probability that the exposure of marsh rice rat to $TEQ_{DF,M}$ exceeds the LOAEL for these effects is 14.3 percent. However, the risks to this receptor are reduced to negligible as a result of implementation of the TCRA.

6.6 Risks to Reptiles

Appendix B describes the literature search for information to support TRVs for reptiles. No information was found to interpret reptile exposures. Extensive literature searches by other authors corroborate this result. Because TRVs needed to interpret exposure estimates for reptiles could not be developed, HQs cannot be calculated, and risks to reptiles cannot be addressed using the same approaches used for other receptors. The risk question presented in Table 3-10, "whether the total daily ingested doses (mg/kg-day) of $COPC_{ES}$ greater than doses known to cause effects on the survival, growth and reproduction of reptiles," cannot be addressed with the available information.

However, exposure estimates for reptiles can be compared to those for other receptors. Table 4-13 shows the CT and RM exposure in mg/kg-day of all wildlife receptors to each $COPC_E$. The estimated daily ingested dose of the alligator snapping turtle is included in this summary. Generally speaking, the estimated exposures to alligator snapping turtle for all of the $COPC_{ES}$ are consistently and substantially lower than for other receptors. This is a reflection of the ingestion rate assumption for the alligator snapping turtle, which is based on the field metabolic rate provided by Nagy et al. (1999). Because reptile metabolic demands are lower than those of birds and mammals, use of an allometric model to estimate ingestion rates, and application of those ingestion rates as the basis for exposure estimates for reptiles, will generally result in lower estimates of ingested doses, assuming reptiles are eating the same types of foods on the Site as birds and mammals.

Because the HQs are generally very low for the other receptors at the Site, this general difference in the level of exposure of reptiles would suggest that risk to reptiles are also negligible for metals, BEHP and PCBs. However, it is not possible to conclude with confidence that risks to alligator snapping turtle and other reptiles from exposure to dioxins and furans are also negligible because risks to molluscs, birds, and mammals from dioxins and furans are present in localized areas adjacent to the northern impoundment, and the relative sensitivity of reptiles to dioxins and furans is unknown. Risk to killdeer from zinc is not an indicator of risk to the alligator snapping turtle because risk to killdeer is a result of exposures originating in soils, and the turtle will be exposed mainly in the aquatic environment.

Uncertainties about risk to reptiles from dioxins and furans also arise from likely differences in the relative importance of the dermal exposure route. Because reptiles lack fur and feathers, and because the skin of some reptiles can have a relatively large lipid content, Weir et al. (2010) have suggested that dermal exposure may be the most important exposure route in reptiles, contributing significantly more of the daily dose of lipid soluble compounds than other exposure routes. There are no means to evaluate this aspect of reptile exposure for the Site, and there are no toxicity data to interpret resulting exposure estimates.

In conclusion, risks to reptiles from metals, BEHP, and PCBs are considered negligible, because risks due to these $COPC_{ES}$ were generally negligible for all other receptors. Even PCBs, which are lipid soluble, are present only at low levels on the Site and are not likely to contribute significantly to reptile risk. Risks to reptiles due to dioxins and furans are unknown, because there are no means to estimate reptile exposures, and no toxicity information to interpret exposures. Because other receptors are exposed to dioxins and furans at levels above those associated with effects in laboratory animals, it may also be true that reptiles using the site at the same frequencies and in the same manner as these other wildlife would have comparable risks.

6.7 Threatened and Endangered Species

Because the evaluation of risk to sandpiper, cormorant, and heron resulted in HQ_N values greater than 1, risks to the white-faced ibis, brown pelican, and bald eagle were evaluated as described in Section 4.3.1.6.

These comparisons were conducted for the white-faced ibis for copper, $TEQ_{DF,B}$, and $TEQ_{DFP,B}$; for brown pelican for $TEQ_{DF,B}$ and $TEQ_{DFP,B}$; and for the bald eagle for $TEQ_{DF,B}$ and

$TEQ_{DFP,B}$. Results are summarized in Table 6-10. No other $COPC_{ES}$ exceeded HQ_N for the surrogate receptors representative of protected species.

Estimated exposures of individuals among protected species that could occur on the Site to $COPC_{ES}$ (i.e., those for which $HQ_N \geq 1$ for the surrogate receptors) do not exceed NOAELs. Therefore, risks to protected species that could occur on the Site are negligible.

6.8 Bioaccumulation and Biomagnification of $COPC_{ES}$

In its comments on the draft BERA (Appendix F), USEPA requires that "the report shall provide/expand its description and evaluation of food chain implications…." Evaluation of bioaccumulation and biomagnification was not included among the DQOs in any of the SAPs for the RI, and data were not collected specifically for that purpose. To effectively evaluate patterns in bioaccumulation or biomagnification using field studies, certain parameters are necessary, such as stable isotopes of nitrogen in various tissue types, including in tissue of primary producers in the study area. Alternatively, controlled experiments can be conducted to evaluate bioaccumulation. None of these types of information are necessary for an RI, and so were not developed for this Site or for this report. To address the USEPA comment for dioxins and furans, a synthesis of information presented in the Technical Memorandum on Bioaccumulation Modeling (Integral 2010b) is presented below. The reader is referred to that report for a detailed discussion.

From the data analyses and the literature review presented in the Technical Memorandum on Bioaccumulation Modeling, including evaluation of region-specific multivariate statistical correlations, it was concluded that the majority of dioxin and furan congeners do not consistently bioaccumulate in fish and invertebrate tissue. Moreover, systematic predictions of bioaccumulation from concentrations of dioxins and furans in abiotic media are difficult and uncertain for some congeners and impossible for others. Uptake efficiencies vary by congener, exposure medium, exposure route, and species. The ability of organisms to transform and eliminate the different dioxin and furan congeners, and the differences in transformation and elimination rates for different congeners adds complexity to patterns of dioxin and furan bioaccumulation across the range of taxa evaluated for this Site. The literature on these subjects is extensive and largely observational. A common conclusion in the literature is that bioaccumulation is controlled more by physiological mechanisms such as the limitations on rates of uptake across gill and gut membranes imparted by the size of dioxin and furan molecules (Opperhuizen and Sijm 1990), and the metabolism and excretion of dioxins and furans, than by chemical properties such as log K_{ow}.

Because rates of uptake and excretion of dioxins and furans are dynamic, species- or taxa-specific, and not described for several congeners, broad generalities are not available to interpret tissue concentrations of dioxins and furans in site-specific samples in terms of the position of each sampled species in the food web. USEPA's (2009) *National Study of Chemical Residues in Lake Fish Tissue* found that benthic fish species overall had higher concentrations of dioxins and furans than predatory fish species, supporting a conclusion that concentrations of dioxins and furans are not predicted by position in the food chain, but are accumulated more as a function of proximity to contaminated sediments. On the Site, whole hardhead catfish have the highest $TEQ_{DF,M}$ concentrations among all tissue types collected for the RI, and hardhead catfish fillet tend to have higher TEQ concentrations than other fish caught on the Site (Exposure Assessment Memo, Appendix B [Integral 2012]), including spotted seatrout and southern flounder, which both eat fish and invertebrates. However, the mean and 95UCL concentrations of $TEQ_{DF,M}$ in whole catfish from FCA2, in which the northern impoundments is located, are the lowest among the three FCAs on the Site. Therefore, results for hardhead catfish suggest that their tissue concentrations of dioxins and furans are higher than for other fish species caught on Site, and more than other species sampled for the RI, but that there is not enough information about the mobility and spatial use patterns, degree of contact with sediment, ages of fish, and other factors to explain the differences. It is notable that clams have the second-highest $TEQ_{DF,M}$ concentrations among tissue collected for the RI on the Site, and that concentrations of individual congeners in clam tissue correlate reasonably well with concentrations in sediments adjacent to where they were collected (PSCR Section 6.2.2.3). Whether clams and catfish occupy similar trophic positions is unknown, but both are more closely associated with the benthic environment than other species for which data are available.

In the absence of specific data to the define trophic structure of the food web on the Site (such as stable nitrogen isotopes or stomach content analysis), no specific conclusions can be drawn about the reasons for higher concentrations of $TEQ_{DF,M}$ in catfish than in other species. Therefore, there are no known "food chain implications" of dioxins and furans in the tissue of species collected for the RI.

7 UNCERTAINTY ANALYSIS

Ecological risk assessments are inherently imprecise and uncertain, and any ecological risk analysis provides only a simplified model of a natural environment that is complex and dynamic. Risk assessors can compensate for uncertainties by using conservative assumptions, but an overly conservative analysis does not effectively inform risk management decisions, and baseline risk assessments should incorporate realism wherever possible. In this section, the following broad categories of uncertainty are described, specific examples from this risk assessment are addressed in detail, and the effects of such uncertainties on the risk evaluation are discussed:

- Data gaps and limitations
- Model uncertainty
- Toxicity information.

Not all of these uncertainties can be addressed by conservatism, so the discussion of each includes a clear statement of whether the resulting bias is conservative, not conservative, or unknown. Finally, several underlying methods and assumptions provide an overall conservatism to the analysis, and these are outlined and described within the categories listed above.

7.1 Data Gaps and Data Limitations

Although a significant number of analytical samples have been collected for the remedial investigation and risk assessments, there are some data gaps that affect the degree of certainty associated with risk estimates: the absence of data for surface water or porewater chemistry; the absence of data for some tissue types that are potentially ingested by receptors; the actual chemical concentrations of sediment on the TCRA cap, now and in the future; a limit to the number of samples that can be collected for the RI; an absence of detailed information about use of the Site by certain protected species; and the lack of information on the toxicity of COPC$_{ES}$ to reptiles. Each of these is discussed below.

7.1.1 Surface Water Chemistry

There are no empirical data in the baseline dataset to describe concentrations of most $COPC_{E}s$ in water, and there are limited water data only for dioxins and furans and PCBs. As a result, a simple model was used to generate conservative estimates of water concentrations for use in the wildlife exposure model, and to estimate porewater chemistry for those chemicals lacking SQGs expressed as bulk sediment concentrations. The results of these simple partitioning models are conservative estimates of $COPC_{E}$ concentrations in water for several reasons. First, the partitioning models represented by the K_d and K_{oc} values used (Table 4-1) assume a two-phase system in equilibrium, and the resulting prediction is for a dissolved concentration. In reality, estuarine surface waters are complex multi-phase systems including several constituents such as dissolved organic carbon and other materials that bind chemicals, preventing them from entering solution. Moreover, it is also unlikely that water and sediment are at equilibrium, given the tidal dynamics in an estuarine environment. Also, because the sediment is in direct contact with only a limited volume of water, most of the surface water would mix with and dilute any metals or other COPCs partitioning from sediment. Such dilution was not accounted for by the simple models used.

The simple models were used because of the absence of empirical data. Because they generate a very conservative representation of water chemistry, they are useful for screening. That is, when these conservative estimates are below levels of concern, the exposure pathway or receptor–COPC pair can be eliminated from further consideration with a high degree of confidence. If an actual estimate of metals or other COPC in surface water were needed, a much larger set of information would be brought to bear, bringing greater realism to the estimate for this particular environment.

7.1.2 Tissue Chemistry for Plants and Terrestrial Invertebrates

Similarly, there are no data to describe concentrations of $COPC_{E}s$ in tissue of terrestrial invertebrates and in plants. For all estimates except for dioxins and furans in terrestrial invertebrate tissue, simple models derived for other sites and published in the literature, or by USEPA or the Oak Ridge National Laboratory for use in risks assessments were applied. These "off-the-shelf" models for estimating plant and invertebrate tissue concentrations provide a reasonable estimate of tissue concentrations, but they cannot account for the Site conditions affecting bioavailability, the particular species studied relative to those on the Site, seasonality of the data providing the basis for the model, local geochemistry and other factors that could affect uptake rates in plants and invertebrates. For the most part, the direction of bias created by using these models is unknown.

7.1.3 Post-TCRA Conditions

To analyze post-TCRA risk, it was necessary to make an assumption about the concentrations and mixture of dioxins and furans in sediments within the area inside the original 1966 impoundment perimeter. To perform the post-TCRA risk analysis, this risk assessment assumes that the area provides the same habitat function that it had up until the TCRA cap was constructed, and that dioxins and furans in sediments within the 1966 perimeter are at the median concentration in the background sediment dataset. This approach assumes that animals will continue to use the area as they did prior to implementation of the TCRA, and that the sediment coming onto the Site, and which becomes deposited on the TCRA cap, is from a broad area similar or equivalent to the background area. The approach also assumes that the conditions remain static, and does not attempt to evaluate the dynamics of sediment deposition and erosion on the TCRA cap in future years. Whether local conditions within USEPA's preliminary Site perimeter would have a greater or lesser effect than the upstream background conditions is unknown. It is also unknown whether the conditions will be static or dynamic, and whether state regulatory programs aimed at controlling releases of dioxins and furans in the region will result in a general lowering of dioxin and furan concentrations in background sediment, which could lower post-TCRA concentrations at the location of the cap.

The selected approach is appropriate because it is not speculative about these details of future conditions. However, if sediment conditions in the area directly adjacent to the 1966 impoundment perimeter do have a disproportionate impact on post-TCRA sediment conditions because of their proximity to the TCRA cap, then the post-TCRA evaluation could slightly underestimate risk. However, the overall conclusion that the TCRA has resulted in significant risk reduction would not change.

7.1.4 Sample Numbers

Designs of all of the studies supporting the RI were developed in collaboration with USEPA, and data collection was performed with USEPA approval of each SAP, with specific DQOs articulated according to the four study elements used to structure the investigation. In each study, the sampling design was directed towards characterizing conditions adjacent to and within the impoundments north of I-10, which is appropriate because the wastes in those impoundments are most likely the primary source of COPCs in the environments addressed by this BERA (a BERA for the southern impoundments is to be presented with the RI). As a result, there is a spatial bias that emphasizes conditions near and in the northern impoundments in the dataset, resulting in a relative over-representation of chemical conditions there in the sediment, soil, and tissue data. This is the most important limitation to the existing data that introduces a conservative bias. Combined with the definition of exposure units that encompass and emphasize the area of the former waste impoundments, the spatially biased sampling designs result in a representation of risk that may overstate exposure and risk to all receptors.

The shoreline exposure unit for great blue heron, spotted sandpiper, and marsh rice rat (Figure 4-14) provides an example of how the spatial bias in the sampling design results in a conservative bias in the exposure analysis. For these receptors, a large proportion of the sediment samples used to calculate EPCs were collected from within the impoundments (more than 50 percent), including several from directly within the wastes, even though the fraction of shoreline on the Site represented by the impoundments is less than 5 percent. Because there was no spatial weighting to normalize the area represented by each sample, the spatial bias in sampling resulting from a focus on the waste impoundments skews the CT and RM exposure statistics upward, directly affecting exposure and risk estimates. Results indicating that implementation of the TCRA resolves risks due to dioxins and furans illustrate the importance of the spatial bias in driving risk estimates.

7.1.5 Threatened and Endangered Species

There are no systematic observations of threatened or endangered species occurring on or near the Site, so use of the Site by those species was inferred on the basis of habitat availability on and near the Site (Section 3.4). As described in Section 3.4.4, among the six threatened or endangered species that could occur in the vicinity of the Site, only the brown pelican would be likely to use the Site for resting and foraging. The bald eagle and white-faced ibis could also visit occasionally. All of these birds have foraging ranges much larger than the Site area, and would therefore be expected to be exposed to a lesser degree than any of the modeled receptors. Evaluation of white-faced ibis, brown pelican, and bald eagle using surrogate receptors and adjustments to exposure areas showed an absence of risk to all three protected bird species.

7.2 Model Uncertainty

Several model forms are used to support the risk assessment, including ratios such as BAFs and BMFs, regression models to predict dioxin and furan concentrations in worm tissue or in bird eggs, and ingested-dose models to estimate exposure of individuals. Each type of model can introduce both bias and inaccuracies.

7.2.1 Prediction Using Ratios

Ratios are the simplest representation of the relationship between chemical concentrations in a source medium, such as sediment or prey, and in tissue. Ratios are calculated as the concentration in the tissue of interest, divided by the concentration in a single exposure medium, which may be sediment, water, or food. Underlying the use of ratios is the assumption of a strictly proportional relationship between concentrations in the two media. Although ratios are widely used, the assumption of proportionality is rarely demonstrated to be justified. For this reason, ratios are the most likely to introduce inaccuracies. Because truly linear relationships between abiotic media or prey and biological tissue are rare, use of ratios introduces a conservative bias, which is made worse at higher concentrations in the abiotic medium. The range and variability of BMFs found linking PCB congeners in foods of fish to those in bird eggs (Table 4-18) illustrates the random variability in predictions that occurs when ratios are used.

7.2.2 Prediction Using Regression Models

Regression analysis of concentrations in tissue and soil or sediment, or between different tissue types (e.g., a consumer and its prey), is a straightforward method using well-established statistical procedures. Regression analysis has several advantages over ratios, specifically the ability to incorporate non-zero intercepts and to produce a statistically sound measure of uncertainty. Because it is a strictly empirical method, regression analysis does not require any information on the mechanisms of exposure and uptake, and thus can be applied to the sort of site characterization data typically collected in an RI/FS. Several guidance documents supporting the use of regression modeling in the process of developing risk assessments and remedial goals have been published (Exponent 1998; Corl 2001; University of Florida 2005). Application of regression models in this risk assessment avoids the sort of overly conservative bias introduced by ratios, because regression models produce predictions within the known variance of the dataset used to build the model. Although Site-specific data are preferred for making predictions for the Site, regression models which limit prediction error on the basis of an empirical dataset are preferred over the use of ratios because they better control the uncertainty in predicted concentrations, at both the lower and upper ends of the predicted range.

7.2.2.1 Fish-to-Bird Egg Models for Dioxins and Furans

Bioaccumulation of dioxins and furans in birds is poorly described in the literature with paucity in available data to guide model development for the prediction of accumulation and transfer from exposure media to tissues of birds. As a result, any estimates of bioaccumulation will result in uncertainties. Use of regression models provides the most straightforward means to limit uncertainty, and requires only the use of data that are available for the Site, instead of multiple variables that would have to be derived from the literature to implement other types of models, such as mechanistic models.

As detailed in the Technical Memorandum on Bioaccumulation Modeling (Integral 2010b), the relationships between exposure concentrations and tissue concentrations in birds is a complex process balancing absorption rates, metabolism, excretion, and maternal transfer, all of which operate on a congener- and species-specific basis. The literature review performed for this risk assessment produced only one study providing a regression based model for estimating concentrations of dioxins and furans in bird eggs (Elliott et al. 2001). Uncertainties associated with this model arise from the derivation of the regression relationship for the individual congener groups. We were unable to directly test whether our input data follows a distribution similar to that of the fish data used to generate the regression. However, visual inspection of a subset of graphically presented data showed a similar distribution of concentrations in prey for PeCDD and HxCDD. Further uncertainty exists for TCDF, which did not result in a strong linear association between prey fish and measured egg concentrations although a positive and potentially nonlinear relationship was evident. Hence, modeled values for this congener are subject to the uncertainty of the regression analysis, which shows that the relationship applies 93 percent of the time (p=0.07) and therefore provides a reasonable level of confidence in the estimated bird egg concentrations. Further, we are unable to assess the uncertainty of congeners for which the regression modeled was derived from other homologous groups (e.g., HpCDD and HpCDF).

Uncertainties also exist in species specific differences in metabolism and excretion of dioxins and furans. The regression model of Elliott et al. (2001) was developed for the great blue heron. This risk assessment applies the models to predict egg concentrations for herons as well as cormorant and sandpipers for which differences in dioxin and furan metabolism may exist. This same difference in metabolism extends to the absorption of compounds from the different tissues. Also, the model was derived using fish as the ingested exposure medium; however, this risk assessment has extended the models to estimate the transfer of dioxins and furans from crab, clam, and sediments. Nosek et al. (1992a) demonstrated differences in the oral bioavailability of TCDD from different matrices in pheasant hens. Similar oral bioavailability was determined for earthworms (30 percent) and soils (33 percent) while higher availability was found for paper mill sludge (41 percent) and a suspension of crickets (58 percent). Application of Elliott et al.'s (2001) regression models to a mixed media diet assumes that the uptake of dioxins and furans from sediments, crab, and clams occurs at the same rate as for those in fish. The importance of this assumption is unknown, but likely somewhat conservative in light of the 30 to 33 percent bioavailability from soils and higher bioavailability from foods demonstrated by Nosek et al. (1992a). However, the assumption was necessary because sediment and shoreline sediment concentrations account for the highest dose of dioxins and furans to birds (Appendix C).

7.2.2.2 Soil-to-Invertebrate Tissue Models for Dioxins and Furans

In modeling dioxin and furan congener uptake into invertebrate tissue, it cannot be assumed that all congeners behave similarly, and a congener-specific approach is needed (Integral

2010a; Appendix D). The regression approach selected to estimate dioxin and furan concentrations in earthworm tissue from concentrations in colocated soils relies on a relatively small sample size (N=6) of colocated soil and earthworm tissue, and there is uncertainty in developing regressions from this small sample size. 2,3,7,8-TCDD was not detected in several of the soil–earthworm pairs in the available dataset, which also introduces uncertainty in the use of these data. However, uncertainties were reduced by limiting selections to statistically significant relationships on a congener-specific basis, and selecting congener correlates to minimize underprediction of tissue concentrations in a soil environment characterized by significant spatial variability in concentrations of some dioxin and furan concentrations. This approach affords substantive advantages in terms of providing an empirically based estimation of invertebrate uptake of dioxins and furans over simplified approaches using biota–sediment accumulation factors or extrapolating widely across aggregate variables such as TEQ.

7.2.3 RM Exposure and the Risk Profile

A CT and RM EPC were generated for each exposure medium within each exposure unit for use in the fish and wildlife exposure models. Using these two expressions of the EPC for any given $COPC_E$ enables presentation of the most likely (CT) exposure, along with the upper bound (RM) exposure condition, and is intended to reflect the "exposure profile" for receptors recommended by USEPA guidance (USEPA 1997, 1998). However, this profile is biased high, because the RMin exposure, calculated as the 95 percent lower confidence limit on the mean (or a similar statistic) is as likely to occur as the RM. An illustration of the importance of this bias in interpreting risks is provided in Table 7-1. This table shows the CT, RM, and RMin $TEQ_{DF,B}$ EPCs for soils used to estimate exposure to killdeer. The RMin is a more than a factor of 7 below the CT. A proportionate decrease in the final HQ_N would lead to a conclusion of negligible risk to killdeer. This example illustrates how the use of the RMin exposure estimate, which is equally as likely as the RM exposure, leads to no finding of risk.

7.2.4 Wildlife Exposure Model

The wildlife exposure model uses fixed parameter values, set at realistic or conservative levels, to make predictions about contaminant intake. Use of conservative assumptions, such as the redistribution of plant matter (likely less contaminated) in the diet of omnivorous benthic fish into a compartment representing animal matter (likely more contaminated) (Table 4-3) is an example of this type of conservatism in the wildlife exposure model. The bird egg exposure models for dioxins and furans also make several conservative assumptions, including the use of total homologue concentrations (not the sum of 2,3,7,8-substituted congeners within a homologue) as the basis for exposure model for several congeners. This approach allows a conclusion of negligible risk to be made with confidence. In some cases, information from the literature that informs specific details of the exposure analysis are employed. While incorporation of this information is considered carefully on the basis of the technical merits of the study or studies reporting the data, it may or may not reflect actual conditions in the field. Examples of this is the use of the RBAs described in Section 4.3.1.2 in the exposure assessment for birds, the choices about proportions of each prey type in the diets of fish and wildlife receptors, and the timing and duration of exposures of each receptor. Generally, these choices are demonstrably conservative; for example, the BERA assumes that blue heron consume a significant amount of catfish, the species with the highest concentrations of dioxins and furans in fillet among those captured on the Site (see Appendix B of the Exposure Assessment Memorandum; Integral 2012), when they likely eat a range of fish sizes and species. The BERA also assumes continuous exposures exclusively in the study area. But not all assumptions are clearly conservative, and are instead included to impart realism to the extent possible. Use of RBAs in the exposure assessment for birds is an example of this.

At USEPA's request in comments on the draft BERA (Appendix F), a sensitivity analysis was conducted to evaluate the importance of the RBAs in risk conclusions for birds. To do this, exposures of birds to dioxins and furans (as $TEQ_{DF,B}$) was recalculated without using RBAs, (i.e., assuming that ingested TCDD was 100 percent bioavailable to the birds). Details of this sensitivity analysis are discussed below.

In addition, a deterministic risk calculation can oversimplify the risk conclusions, by suggesting a black and white risk/no risk conclusion. To improve the depth of the evaluation, risk was evaluated probabilistically when the deterministic models suggested that exposures could exceed the LOAEL. Use of the probabilistic food web exposure model is also discussed below.

7.2.4.1 Alternative Assessment of Exposure of Birds to for Dioxins and Furans

A sensitivity analysis was conducted to evaluate the importance of the RBAs in risk conclusions for birds. To do this, exposures of birds to dioxins and furans (as $TEQ_{DF,B}$) were recalculated without using RBAs (i.e., assuming that ingested TCDD was 100 percent bioavailable to the birds). The results of these analyses are presented in Table 7-2. Tables 7-3 and 7-4 also provide a comparison of the effect of the RBA on pre-and post-TCRA analyses and site and background analyses, respectively. Probabilistic risk analyses of sandpiper and killdeer exposure to $TEQ_{DF,B}$ without the RBA are provided in Figures 7-1 and 7-2.

The results of these analyses are similar to results with the RBA; the probability that exposures of killdeer to dioxins and furans will exceed the effects level is still low (5.6 percent without the use of the RBA [Figure 7-1] compared to 4.7 percent with the RBA [Figure 6-19]), and the probability that the exposures of spotted sandpiper to dioxins and furans will exceed the effects level is still low (14.7 percent without the use of the RBA [Figure 7-2] compared to 13.7 percent with the RBA [Figure 6-20]). There are no changes to the risk conclusions relative to the outcome of the risk analysis using the RBA. Thus, while the use of the RBA is considered an appropriate adjustment to TCDD bioavailability, its use does not have a substantive effect on the outcome of the risk evaluation for avian receptors.

7.3 Use of Probabilistic Exposure Models

In this risk assessment, when predicted ingestion exposures exceeded LOAELs, the probabilistic exposure assessment incorporates the variability of the exposure parameters in the deterministic model, providing a more precise statement of probability of adverse outcomes (e.g., an 8.3 percent chance that exposure of killdeer to zinc will exceed the LOAEL). This statement uses empirical information about the Site to more accurately reflect likelihood of an effect on an individual.

7.4 Toxicity Information

Ecological risk assessments rely on a very limited set of toxicity information, usually developed with very few species derived from domestic stocks. Often, these domestic species are less fit than wild species, which benefit from greater genetic diversity in each generation.

The advantage of using controlled laboratory studies is certainty in the dose-response relationship that is derived from highly controlled exposures such as injection or oral administration by gavage. However, loss of realism is significant to risk assessment, because toxicity studies cannot represent variability in individual fitness, variable resistance or sensitivity among species, physical controls on bioavailability that exist in the field, and a host of other factors affecting potential toxicity in the environment. Generally, the bias resulting from the use of laboratory-based toxicity studies is considered conservative.

Toxicity information for PCBs in fish, birds, and mammals relied largely on toxicity studies in which the test subjects were administered Aroclor 1254. To achieve a risk assessment protective of the ecological receptors addressed by this BERA, concentrations of total PCBs were used in exposure estimates. For sediments, PCB congener data are insufficient for calculating total PCBs as the sum of congeners. The concentrations of total PCBs in sediment were therefore estimated by summing the concentrations of Aroclors with nondetects set to one-half the detection limit. Because data for PCB congeners are available for tissue samples (and Aroclor data are not), total PCBs in prey was estimated by summing the concentrations of 43 congeners analyzed in prey tissue.

Finally, very conservative assumptions were applied in calculating total PCBs in sediment as the sum of Aroclors with nondetects set to one-half the detection limit. For the sediment study, PCBs were analyzed and reported as Aroclors, consistent with the Sediment SAP.[5] However, in several samples of material from within the 1966 impoundment perimeter, matrix interferences resulted in elevated detection limits for Aroclors. The use of these elevated detection limits for the sum of Aroclors likely results in a substantial overestimate of the sediment EPC for total PCBs. This is a conservative assumption because no Aroclors were detected in surface sediment within the 1966 impoundment perimeter during the sediment study for this RI, and only a single detected concentration of Aroclor 1254 was measured at depth (2 to 4 feet) within this area (i.e., Station SJGB014, 1,400 µg/kg [qualifier – J]). This estimated concentration is lower than the elevated detection limit for this Aroclor in two of the stations where detection limits were elevated. Moreover, in the Screening Site

[5] The USEPA comment requiring evaluation of exposures to total PCBs as the sum of 43 specific congeners was first articulated in the comments on the Tissue SAP, which was produced after the Sediment SAP was final and implemented. See Appendix C of the Tissue SAP.

Assessment Report (TCEQ and USEPA 2006), which reports Aroclor results for several samples from within the wastes in the western cell of the northern impoundments, Aroclors were never detected, and detection limits were much lower in that study (<90 µg/kg). In summary, there is uncertainty about the actual Aroclor concentrations in the materials collected from within the 1966 impoundment, but the estimated concentration of Aroclor 1254 at Station SJGB014, and results of TCEQ and USEPA (2006) confirm that the approach taken to estimating total PCBs in sediment is conservative.

In addition to uncertainty in calculation of sediment EPCs for total PCBs, there is uncertainty associated with the use of Aroclor 1254 toxicity information in combination with total PCBs as the exposure metric. The mixture of PCB congeners in sediments and tissue at the Site may not reflect the same congener composition as Aroclor 1254. Nevertheless, the assessment approach should be protective because Aroclor 1254 is expected to be among the Aroclors most toxic to aquatic organisms (Nebeker and Puglisi 1974; Mayer et al. 1977; Johnson and Finley 1980), and dechlorination of PCBs by natural processes at the Site would likely lead to mixtures with toxicity less than or equal to Aroclor 1254.

7.4.1 Use of Uncertainty Factors for Deriving TRVs

The preferred approach for selecting TRVs is to find values that meet acceptability criteria (Section 1.4 of Appendix B) and are taxonomically relevant and appropriate to the receptors of concern, but in some cases, data may not be available for the receptor and COPC of interest. In these cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV may be considered. In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk.

Chapman et al.'s (1999) review emphasizes the importance of evaluating the substance and context of the uncertainty. They caution against the extrapolation of LOAELs to NOAELs because there can be substantial uncertainty in moving from effects to no-effects concentrations. They provide several examples that support the use of uncertainty factors of 10 or less for individual extrapolations, including extrapolation of acute lethality toxicity tests to thresholds for sublethal effects in aquatic systems, and lowest-observed-effect concentration to no-observed-effect concentration ratios for wildlife criteria (Chapman et al. 1999). This review points out that uncertainty factors are essentially screening tools for which the imprecision cannot be quantified, and should not be regarded as mathematical absolutes. These recommendations were used as a basis for the application of uncertainty factors in deriving TRVs where relevant effects level values were missing but related values were available.

7.4.2 Toxicity of Mixtures

Organisms inhabiting or using the Site are exposed to more than one chemical. They may be exposed to COPCs and other chemicals in locations other than the Site. Exposures of organisms to chemicals other than COPCs off of the Site cannot be estimated. Each contaminated area and each individual receptor results in a unique exposure profile, characterized by both the specific chemical mixtures and the magnitude of each chemical. For most chemicals, there are simply no published studies to evaluate these unique mixtures.

Some chemicals have known additivity, such as dioxins and furans, and mixtures are evaluated on the basis of the best available science. For other COPCs, whether effects associated with one COPC are additive with another cannot be addressed without significant effort, and may not be resolved in any case. In this evaluation, because very few COPCs are associated with unacceptable risk, evaluation of mixtures other than dioxin, furan, and dioxin-like PCBs was not conducted. The related uncertainty is considered minor for this Site.

7.4.3 Bivalve Toxicity for Dioxins and Furans Other than TCDD

Appendix B provides an overview of the technical literature available for the evaluation of dioxin and furan toxicity to invertebrates. Several studies have found no adverse effects in freshwater or marine invertebrates following exposure to TCDD; studies to provide systematic toxicity data for the other dioxin and furan congeners are rare. The literature and related analyses find that invertebrates are relatively insensitive to TCDD toxicity. Although AhR homologues have been identified in various invertebrate species, invertebrate AhR homologues lack the ability to bind dioxins (Hahn et al. 1992; Butler et al. 2001), which may explain the relatively low sensitivity of invertebrates to TCDD toxicity.

The histological and developmental toxicity of TCDD in eastern oysters documented by Wintermyer and Cooper (2007) illustrates that TCDD toxicity to bivalves is likely through non-AhR mediated pathways. Therefore, this study and related literature do not inform the question of whether other dioxin and furan congeners may also have similar effects in oysters, other additional effects, or no effects at all. Because the mechanism of toxicity to oyster reproductive tissues is not described, it cannot be concluded that other dioxin and furan congeners may have the same effects. Because these other congeners have not been tested, it cannot be concluded that they do not have any effect. Therefore, the absence of information on the toxicity of dioxin and furan congeners other than TCDD results in uncertainty about risks to bivalves on the Site from these chemicals.

7.4.4 Toxicity to Reptiles

Finally, the absence of information on toxicity of COPCEs to reptiles and the inability to estimate actual exposures to reptiles, which may include dermal uptake, create important uncertainties. Even if reptile tissue samples from the Site had been collected, interpretation of these in terms of risks would not be possible. Calculation of exposure in terms of daily ingested dose for each COPCE, and comparison with birds and mammals, which generally showed little to no risks for the majority of COPCEs, suggest that there are no risks to reptiles for most COPCEs. However, a remaining uncertainty that cannot be resolved is whether baseline exposures of reptiles to dioxins and furans were at levels that could result in unacceptable risks.

7.5 Summary of Uncertainties

Uncertainty in an ecological risk assessment is unavoidable, because the risk assessment attempts to model the natural environment, which is highly variable and complex. Although a baseline ecological risk assessment should incorporate realism to the maximum extent possible, conservative choices are made throughout the process, making the risk assessment generally conservative. Even so, data gaps such as the lack of toxicity information for reptiles, and randomness, such as that introduced by use of ratios to make predictions, cannot be resolved, and their bias cannot be said to be conservative or otherwise. The analyses presented here result in a high level of confidence when there is a conclusion of no risk.

8 SUMMARY OF ECOLOGICAL RISKS AND RISK CONCLUSIONS

This section synthesizes the results of the risk characterization and uncertainty analysis to provide an overall conclusion about the ecological risks at the Site.

8.1 Characterization of Risks to Benthic Invertebrates

A conservative assessment of risks to benthic invertebrates indicates no risks to the assessment endpoint of the abundance and diversity of benthic macroinvertebrate communities from exposure to BEHP, phenol, cobalt, copper, lead, thallium, and zinc. Carbazole and aluminum concentrations in surface sediments of the Site are not greater than in background areas, and risks associated with these metals are therefore not greater than background risks. Barium and vanadium, for which information on toxicity to benthic macroinvertebrates is lacking, and manganese are randomly distributed in sediments, and therefore appear not to be associated with the source material in the impoundments. Concentrations of mercury exceed a conservative SQG in two locations, but these exceedances do not equate to a prediction of effects. If effects exist at these two locations, the affected areas are isolated and small, and do not adversely affect the assessment endpoint, abundance and diversity of the overall benthic macroinvertebrate community.

Concentrations of TCDD in sediment exceed the NOAEL in only two locations, within the original impoundment perimeter, but there were no studies identifying benthic invertebrate LOAELs for dioxins and furans in sediment. NOAEL values as high as 25,000 ng/kg have been reported (Appendix Table B-4), suggesting that concentrations of TCDD in sediments are not sufficiently high to negatively impact the benthic macroinvertebrate community.

Clam tissue concentrations of TCDD are sufficiently elevated in samples collected directly adjacent to the impoundments to indicate reproductive risks to individual molluscs in that area. Concentrations of TCDD in clam tissue from two of five samples at Transect 5, directly adjacent to the upland sand separation area, exceed a threshold of reproductive effects in individual oysters. These localized effects do not adversely affect the assessment endpoint, stable or increasing populations of bivalves within the Site, because the affected area is limited to the immediate vicinity of the impoundments north of I-10.

8.2 Characterization of Risks to Fish

Assessment of baseline risks to fish considered the concentrations of cadmium, copper, mercury, and zinc in the diets of fish, the concentrations of BEHP and nickel in water, and the concentrations of total PCBs, $TEQ_{DF,F}$, $TEQ_{P,F}$, and $TEQ_{DFP,F}$ in whole fish. Results indicate that baseline risks to the assessment endpoint, stable or increasing populations of benthic omnivorous fish, benthic invertivorous fish, and benthic piscivorous fish on the Site are negligible.

8.3 Characterization of Risks to Birds

Baseline risks to the assessment endpoint of stable or increasing populations of great blue heron and neotropic cormorant, and the birds in their feeding guilds that are represented by these receptor surrogates and that could use the Site are negligible. Exceedance of the egg tissue based NOAEC for great blue heron and cormorant ingesting prey and sediment at the Site are noted, but do not indicate risk to the assessment endpoints for piscivorous birds. Baseline risks to terrestrial invertivorous birds such as the killdeer are also negligible for all $COPC_{ES}$ except zinc and dioxins and furans. Baseline risks to spotted sandpiper and similar shorebirds, which ingest substantial amounts of sediment as a result of their foraging habit, are negligible for all $COPC_{ES}$ except for dioxins and furans.

There is a low probability (8.3 percent) that exposures of individual killdeer to zinc could exceed levels affecting reproduction, indicating negligible risk to the assessment endpoint of stable or increasing populations of terrestrial invertivorous birds. Uncertainties about the bioavailability of zinc from site soils, and of the form of this metal in foods and soils on the Site relative to the form used in toxicity tests result in a conservative bias in the risk assessment for zinc in killdeer. Exposures of killdeer to zinc on the Site are only slightly greater than exposures in background areas. There is also a low probability (4.7 percent) that exposures of individual killdeer to $TEQ_{DF.B}$ could exceed the LOAEL at the Site. Overall, risks to terrestrial invertivorous bird populations on the Site from zinc are very low to negligible.

There is a probability of 13.7 percent that exposure of individual spotted sandpipers and the species it represents to dioxins and furans exceeds exposures associated with reproductive effects in individual birds under baseline (pre-TCRA) conditions. Although probability of this exposure level was only calculated using the ingestion rate of birds, results of the modeling to estimate egg concentrations also indicate some baseline risk of reproductive effects from dioxins and furans in the spotted sandpiper. Among all vertebrate ecological receptors for this risk assessment, the sandpiper ingests the largest amount of sediment (per unit body weight), which is the most important source of their exposure. Implementation of the TCRA reduced risk to spotted sandpiper to negligible.

8.4 Characterization of Risks to Mammals

Baseline risks to raccoon and mammals in the same feeding guild as the raccoon that could use the Site are negligible. There is negligible risk to the assessment endpoint of stable or increasing populations of omnivorous mammals from any $COPC_E$. Baseline risks to the marsh rice rat, representative of aquatic mammals, are also negligible for all $COPC_{ES}$ except dioxins and furans. There is a 14.3 percent probability that an individual marsh rice rat using the Site under baseline conditions could be exposed to $TEQ_{DFP,M}$ at levels exceeding those associated with reproductive effects on mammals. Given the spatial bias in the dataset towards areas containing the most contaminated sediment on the Site, and given that these rodents can rear more than one litter each year (Appendix A), and that the probability of exposure at the effects level is low, baseline risks to the assessment endpoint of stable or increasing populations of omnivorous mammals on the Site as a whole are negligible. Implementation of the TCRA eliminated risks to the marsh rice rat and the mammals it represents.

8.5 Characterization of Risks to Reptiles

There is insufficient information on the toxicity of $COPC_{ES}$ to specifically address risks to the assessment endpoint of stable or increasing populations of reptiles using the Site. Although there are substantial uncertainties about dermal absorption of $COPC_{ES}$, in addition to uncertainties about toxicity, comparison of the alligator snapping turtle's ingested doses with those of bird and mammal receptors indicates that exposure potential of reptiles via ingestion is very low. For this reason, and because risks to $COPC_{ES}$ other than dioxins and furans are low for some but more often negligible for these other receptors, risks to reptiles to $COPC_{ES}$ other than dioxins and furans are also considered to be low. However, risks to reptiles living in close association with the former waste impoundments from exposure to dioxins and furans could exist under baseline conditions, because risks to spotted sandpiper and marsh rice rat are present, and because reptiles may be more susceptible to dermal uptake of dioxins and furans, increasing their exposure over estimates presented herein. Similarly, because implementation of the TCRA resolves risks to sandpiper and marsh rice rat, any risk to reptiles, if present, would be similarly reduced. Risks to reptiles from exposure to dioxins and furans are unknown.

8.6 Ecological Risk Assessment Conclusions

Baseline risks to benthic macroinvertebrate communities and populations of fish, birds, mammals, and reptiles resulting from the presence of metals, BEHP, PCBs, carbazole, and phenol on the Site are negligible. Risks to fish populations from all COPCEs are negligible. There are negligible risks to populations of wading birds represented by the great blue heron, and to populations of diving birds like the neotropic cormorant. There are negligible risks to populations of terrestrial mammals such as the raccoon.

There are low to negligible risks to individual terrestrial invertivorous birds like the killdeer from exposure to zinc, and negligible risks to populations of such birds. Although the upper bound of estimated daily intakes of zinc by individual killdeer is about equal to conservative effects thresholds, the exposure estimate is influenced by the use of generic models to estimate zinc concentrations in the foods of the killdeer, and this model likely overestimates ingested tissue concentrations, resulting in overestimates of exposure and risk. The highest exposures of killdeer to zinc occur outside of the northern impoundment perimeter, and background exposures less than 30 percent lower than on the Site. In addition, the low probability of individual exposures exceeding effects levels indicates low risk to populations. There are also low to negligible risks to individual terrestrial invertivorous birds from exposure to dioxins and furans.

Baseline risks to ecological receptors associated with the wastes in the impoundments north of I-10 are the result of exposures to dioxins and furans localized to the immediate vicinity of the impoundments. Baseline ecological risks include reproductive risks to molluscs from exposure to TCDD, but primarily in the area of Transect 3, which surrounds the former waste impoundments, and low risks of reproductive effects in individual molluscs in sediments adjacent to the upland sand separation area, but not to populations of molluscs. Baseline risks include moderate risks to individual birds like the killdeer or spotted sandpiper whose foraging area could regularly include the shoreline adjacent to the impoundments north of I-10, but low risk to populations because of the low to moderate probability that individual exposures reach effects levels. Baseline risks include risks to individual small mammals with home ranges that include areas adjacent to the impoundments such as the marsh rice rat, but low to negligible risks to small mammal populations because of the moderate probability that exposures will reach levels associated with reproductive effects in individuals, and because small mammals reproduce rapidly.

To the extent that risks from chemicals other than dioxins and furans occur on the Site, they are not associated solely with hazardous substances that may have been released from the wastes in the former impoundments. Substantial exposure of killdeer to zinc, and a variable fraction of the exposures of several receptors to PCBs, occur in background areas.

Implementation of the TCRA has reduced individual and population-level risks associated with dioxins and furans to negligible, but does not affect risks to killdeer from zinc, suggesting that the wastes in the northern impoundments are not the primary source of exposures of killdeer to zinc. Results of the evaluation of post-TCRA ecological risks support the conclusion that localized exposures of ecological receptors to the wastes in the northern impoundments is the primary driver of baseline ecological risk at the Site, and that therefore risks are localized, resulting from direct contact with the wastes in the northern impoundments.

9 REFERENCES

Alexander, G.R., 1997. Food of vertebrate predators on trout waters in north central lower Michigan. *Mich. Academ.* 10:181–195.

American Society of Mammalogists, 2009. Mammals of Texas. Accessed at http://www.mammalsociety.org/mammals-texas.

Anchor QEA and Integral, 2010. Final Remedial Investigation/Feasibility Study Work Plan, San Jacinto Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. November 2010.

Anchor QEA, 2011a. Draft Removal Action Completion Report, San Jacinto Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS. October 2011.

Anchor QEA, 2011b. Addendum 1 to the Groundwater Study Sampling and Analysis Plan (SAP) for additional groundwater sampling south of Interstate Highway 10 (I-10), San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS. December 2011.

Anchor QEA, 2012. Chemical Fate and Transport Modeling Report San Jacinto Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS. February 2012

Bacci, E., M.J. Cerejeria, C. Gaggi, G. Chemello, D. Calamari, and M. Vighi, 1992. Chlorinated dioxins: volatilization from soils and bioconcentration in plant leaves. *Bull. Environ. Contam. Toxicol.* 48(3):401-408.

Becker, D.S., and T.C. Ginn, 2008. Critical evaluation of the sediment effect concentrations for polychlorinated biphenyls. *Integ. Environ. Assess. Manage.* 4(2):156-170.

BESI, 2010. San Jacinto Waste Pits, Wetland Determination Report, Harris County, Texas. Prepared for Anchor QEA, LLC. Benchmark Ecological Services, Inc. November

Beyer, W.N., E.E. Connor, and S. Gerould, 1994. Estimates of soil ingestion by wildlife. *J. Wildl. Manage* 58(2):375–382.

Birnbaum, L.S., 1994. The mechanism of dioxin toxicity: Relationship to risk assessment. *Environ. Health Perspect.* 102(Suppl 9):157-167.

Bishop, C.A., R.J. Brooks, J.H. Carey, P. Ng, R.J. Norstrom, and D.R.S. Lean, 1991. The case for a cause-effect linkage between environmental contamination and development in eggs of the common snapping turtle (*Chelydra S. serpentina*) from Ontario, Canada. *J. Toxicol. Environ. Health.* 33:521–547.

Braune, B.M., and R.J. Norstrom, 1989. Dynamics of organochlorine compounds in herring gulls: III. Tissue distribution and bioaccumulation in Lake Ontario gulls. *Environmental Toxicology and Chemistry* 8:957-968.

Broach, L., 2010. Personal communication (email exchange between D. Rudnick, Integral Consulting Inc. and L. Broach, TCEQ, discussing molluscs collected from the vicinity of the San Jacinto Waste Pits Site). January 13, 2010.

Broman, D., C. Naf, C. Rolff, Y. Zebuhr, B. Fry, and J. Hobbie, 1992. Using ratios of table nitrogen isotopes to estimate bioaccumulation and flux of polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) in two food chains from the northern Baltic. *Environ. Toxicol. Chem.* 11:331–345.

Bromilow, R.H., and K. Chamberlain, 1995. Principles governing uptake and transport of chemicals. In: Plant Contamination Modeling and Simulation of Organic Chemical Processes. S.Trapp, J.C.McFarlone (eds). Lewis Publisher, Boca Raton, FL.

Budinksy, R.A., J.C. Rowlands, S. Casteel, G. Fent, C.A. Cushing, J. Newsted, J.P. Giesy, M.V. Ruby, and L.L. Aylward, 2008. A pilot study of oral bioavailability of dioxins and furans from contaminated soils: Impact of differential hepatic enzyme activity and species differences. *Chemosphere* 70:1774-1786.

Burton, D.T., S.D. Turley, D.J. Fisher, D.J. Green, and T.R. Shedd, 2006. Bioaccumulation of total mercury and monomethylmercury in the earthworm *Eisenia fetida*. *Water Air Soil Pollut.* 170:37-54.

Butler, R.A., M.L. Kelley, W.H. Powell, M.E. Hahn, and R.J. Van Beneden, 2001. An aryl hydrocarbon receptor homologue from the soft-shell clam, *Mya arenaria*: evidence that invertebrate AHR homologues lack TCDD and BNF binding. *Gene* 278:223-234.

Butler, R.A., M.L. Kelley, K.E. Olberding, G.R. Gardner, and R.J. Van Beneden, 2004. Aryl hydrocarbon receptor (AhR)-independent effects of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) on softshell clam (*Mya arenaria*) reproductive tissue. Comparative *Biochemistry and Physiology*, Part C, Toxicology & Pharmacology 138(3):375-381.

Calder, W.A., and E.J. Braun, 1983. Scaling of osmotic regulation in mammals and birds. *Am. J. Physiol.* 244:601–606.

Carney, S.A., A.L. Prasch, W. Heideman, and R.E. Peterson, 2006. Understanding dioxin developmental toxicity using the zebrafish model. *Birth Defects Research* 76(1):7-18.

Chapman, P.M., A. Fairbrother, and D. Brown, 1999. A critical evaluation of safety (uncertainty) factors for ecological risk assessment. *Environ. Toxicol. Chem.* 17(1):99-108.

Clark, F.W., 1924. The Data of Geochemistry 5. USGS Bulletin 770.

Clark, R.D., T.J. Minello, J.D. Christensen, P.A. Caldwell, M.E. Monaco, and G.A. Matthews, 1999. Modeling Nekton Habitat Use in Galveston Bay, Texas: An Approach to Define Essential Fish Habitat (EFH). NOAA/NOS Biogeography Program, Silver Spring, MD, and NMFS, Galveston, TX. 70 pp.

Cohen-Barnhouse, A.M., M.J. Zwiernik, J.E. Link, S.D. Fitzgerald, S.W. Kennedy, J.P. Giesy, S. Wiseman, P.D. Jones, J.L. Newsted, D. Kay, and S.J. Bursian, 2011. Developmental and posthatch effects of *in ovo* exposure to 2,3,7,8-TCDD, 2,3,4,7,8-PeCDF, and 2,3,7,8-TCDF in Japanese quail (*Coturnix japonica*), common pheasant (*Phasianus colchicus*), and white leghorn chicken (*Gallus gallus domesticus*) embryos. *Environmental Toxicology and Chemistry* 30(7):1649-1668.

Cooper, K.R., and M.L. Wintermyer, 2009. A critical review: 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) effects on gonad development in bivalve mollusks. *Journal of Environmental Science and Health* Part C, Environmental, Carcinogenesis & Ecotoxicology Reviews 27(4):226-245.

Corl, E., 2001. Bioaccumulation in the Ecological Risk Assessment (ERA) Process. Issue Paper, Navy Guidance for Conducting Ecological Risk Assessments. Available at: http://web.ead.anl.gov/ecorisk/issue/. Naval Facilities Engineering Command, Atlantic Division. August 7, 2001. 12 pp.

Crocker, P.A., and C. Young, 1990. Tetrachlorodibenzo-p-dioxins and dibenzofurans in edible fish tissue at selected sites in Arkansas, Louisiana and Texas. U.S. Environmental Protection Agency, Region 6, Water Quality Management Branch, Technical Section, 1445 Ross Avenue, Dallas, TX 75202-2733.

Culbertson, J., 2010. Personal communication (email exchange between D. Rudnick, Integral Consulting Inc. and J. Culbertson, Texas Parks and Wildlife Department, discussing molluscs likely to be found in upper Galveston Bay. January 12, 2010.

Custer, C.M., and J. Galli, 2002. Feeding habitat selection by great blue herons and great egrets nesting in east central Minnesota. *Waterbirds* 245: 115-124.

Custer, C.M., T.W. Custer, C.J. Rosiu, M.J. Melancon, J.W. Bickham, and C.W. Matson, 2005. Exposure and effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in tree swallows (*Tachycineta bicolor*) nesting along the Woonasquatucket River, Rhode Island, USA. *Environmental Toxicology and Chemistry* 24(1):93-109.

Davis, W.B., and D.J. Schmidly, 1994. The Mammals of Texas: Online Edition. Accessed at http://www.nsrl.ttu.edu/tmot1/Default.htm. Accessed on September 27, 2011.

DREBWQAT, 1999. King County Combined Sewer Overflow Water Quality Assessment for the Duwamish River and Elliott Bay. Appendix B: Methods and Results, B3: Wildlife Risk Assessment. Prepared by the Duwamish River and Elliott Bay Water Quality Assessment Team. King County Department of Natural Resources, Seattle, WA.

Eadon, G., L. Kaminsky, and J. Silkworth, 1986. Calculation of 2,3,7,8-TCDD equivalent concentrations of complex environmental contaminant mixtures. *Environ Health Perspect.* 70:221-227.

Elliott, J.E., M.L. Harris, L.K. Wilson, P.E. Whitehead, and R.T. Norstrom, 2001. Monitoring temporal and spatial trends in polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) in eggs of great blue heron (*Ardea herodias*) on the Coast of British Columbia, Canada, 1983-1998. *Ambio.* 30(7):416-428.

Elonen, G.E., R.L. Spehar, G.W. Holcombe, R.D. Johnson, J.D. Fernandez, R.J. Erickson, J.E. Tietge, and P.M. Cook, 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin to seven freshwater fish species during early life-stage development. *Environmental Toxicology and Chemistry* 17:472-483.

Elsey, R.M., 2006. Food habits of *Macrochelys temminckii* (alligator snapping turtle) from Arkansas and Louisiana. *Southeastern Naturalist* 5(3):443-452.

Erickson, M.D., 1997. *Analytical Chemistry of PCBs*. Second Edition. Lewis Publishers.

Exponent, 1998. Review of Bioaccumulation Methods for Fish and Shellfish. Prepared for the Washington State Department of Ecology, Olympia, WA. Exponent Inc., Bellevue, WA. December 1998.

Fengxiang, X.H., Y. Su, Z. Shi, Y. Xia, W. Tian, V. Philips, DL. Monts, M. Gu, and Y. Liang, 2012. Mercury distribution and speciation in floodplain soils and uptake into native earthworms (*Diplocardia* spp.). *Geoderma* 170:261-268.

Fernandes, M.V., 2011. Effects of Changes in the Everglades on Two Indicator Species: *Sigmodon hispidus* and *Oryzomys palustris*. Open Access Dissertations. Paper 606. http://scholarlyrepository.miami.edu/oa_dissertations/606

Fox, G.A., 2001. Wildlife as sentinels of human health effects in the Great Lakes-St. Lawrence Basin. *Environ Health Perspect* 109(Suppl 6):853-861.

Fox, P.M., and D.E. Doner, 2003. Accumulation, release, and solubility of arsenic, molybdenum, and vanadium in wetland sediments. *J. Environ Qual.* 32(6):2428-35.

Frank, D.S., M.A. Mora, J.L. Sericano, A.L. Blankenship, K. Kannan, and J.P. Giesy, 2001. Persistent organochlorine pollutants in eggs of colonial waterbirds from Galveston Bay and East Texas, USA. *Environmental Toxicology and Chemistry* 20(3):608-617.

Gardiner, J., B. Azzato, and M. Jacobi (eds.), 2008. Coastal and Estuarine Hazardous Waste Site Reports, September 2008. Seattle: Assessment and Restoration Division, Office of Response and Restoration, National Oceanic and Atmospheric Administration. 148 pp.

GBIC, 2009. Shrimp and Crabs. Available at: http://gbic.tamug.edu/ss/pubs/shrimpcrabs.pdf Accessed on January 7, 2010. Galveston Bay Information Center.

Gentry, B., E. Wainwright, and D. Blankinship, 2005. Crystal Ball® 7.1 user manual. Decisioneering, Inc., Denver, CO. 357 pp.

Gonzalez, L.A., J.P. DallaRosa, and L. Robinson, 2006. Invasive Species Initiatives in the Galveston Bay Estuary: Risk Assessment, Research, Management and Outreach. Accessed at http://www.icais.org/pdf/2006ppt/Gonzalez_Lisa.pdf. Accessed on January 7, 2010.

Hahn, M.E., A. Poland, E. Glover, and J.J. Stegeman, 1992. The Ah receptor in marine animals: Phylogenetic distribution and relationship to Cytochrome P4501A inducibility. *Marine Environmental Research* 34:87-92.

Hassan-Williams, C., C.H. Bonner, and C. Thomas, 2012. Texas Freshwater Fishes. Texas A&M Press. Available at http://www.bio.txstate.edu/~tbonner/txfishes/index.htm.

Heinz, G.H., 1979. Methyl mercury: reproductive and behavioral effects on three generations of mallard ducks. *J. Wildl. Manage.* 43:394–401.

Helsel, D.R., 2005. Nondetects and data analysis: statistics for censored environmental data. John Wiley & Sons, Inc., Hoboken, NJ.

Henshel, D.S., B. Hehn, R. Wagey, M. Vo, and J.D. Steevens, 1997a. The Relative Sensitivity of Chicken Embryos to Yolk- or Air-Cell-Injected 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin. *Environmental Toxicology and Chemistry* 16(4):725-732.

Hu, K., and N.J. Bunce, 1999. Metabolism of polychlorinated dibenzo-*p*-dioxins and related dioxin-like compounds. *J. Toxicol. Environ. Health*, Part B 2(2):183–210.

Hu, X., B. Wen, S. Zhang, and X. Shan, 2005. Bioavailability of phthalate congeners to earthworms (*Eisenia fetida*) in artificially contaminated soils. *Ecotoxicology and Environmental Safety* 62: 26-34.

Integral, 2010a. Sampling and Analysis Plan (SAP): Soil Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

Integral, 2010b. Technical Memorandum on Bioaccumulation Modeling, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance

Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September 2010.

Integral, 2011a. Soil Sampling and Analysis Plan, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. January 2011.

Integral, 2011b. Draft Addendum 2 to the Sediment Sampling and Analysis Plan (SAP) for Additional Sediment Sampling South of Interstate Highway 10 (I-10), San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. December, 2011.

Integral, 2011c. COPC Technical Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. May 2011.

Integral, 2011d. Draft Addendum 3 to the Soil Sampling and Analysis Plan (SAP) for additional Soil Sampling South of Interstate Highway 10 (I-10), San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. December, 2011.

Integral, 2011e. Field Sampling Report: Tissue Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September.

Integral, 2012. Exposure Assessment Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. May 2012.

Integral and Anchor QEA, 2010. Sampling and Analysis Plan: Sediment Study San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection

Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. April 2010.

Integral and Anchor QEA, 2011a. Field Sampling Report: 2010–2011 Soil Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA, and Anchor QEA, LLC, Ocean Springs, MS. July.

Integral and Anchor QEA, 2011b. Field Sampling Report: 2010 Sediment Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA, and Anchor QEA, LLC, Ocean Springs, MS. July.

Integral and Anchor QEA, 2012. Preliminary Site Characterization Report, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. February, 2012.

Hahn, M.E., A. Poland, E. Glover, and J.J. Stegeman, 1992. The Ah receptor in marine animals: Phylogenetic distribution and relationship to cytochrome P4501A inducibility. *Marine Environmental Research* 34:87-92.

Heiden, T.K., R.J. Hutz, and M.J. Carvan, III, 2005. Accumulation, Tissue Distribution, and Maternal Transfer of Dietary 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin: Impacts on reproductive success of zebrafish. *Toxicological Sciences* 87(2):497-507.

Jackson, B.J., and J.A. Jackson, 2000. Killdeer *(Charadrius vociferus)*, The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/517.

Jarman, W.M., W.J. Sydeman, K.A. Hobson, and P. Bergqvist, 1997. Relationship of polychlorinated dibenzo-*p*-dioxin and polychlorinated dibenzofuran levels to stable-nitrogen isotope abundance in marine birds and mammals in coastal California. *Environ. Toxicol. Chem.* 16(5):1010–1013.

Johnson, W.W., and M.T. Finley, 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Summaries of toxicity tests conducted at Columbia National Fisheries Research Laboratory, 1965-78. U.S. Fish Wildl. Serv. Resour. Publ. 137. 98p.

Kabata-Pendias, A., and H. Pendias, 1992. Trace elements in Soils and Plants. 2nd edition. CRC Press, Inc., Boca Raton, FL.

King, K.A., 1989. Food habits and organochlorine contaminants in the diet of olivaceous cormorants in Galveston Bay, Texas. Southwest. *Nat.* 34:338-343.

Koenig, L., 2010. Personal Communication (telephone conversation with D. Rudnick, Integral Consulting Inc., Seattle, WA, on March 12, 2010, regarding sediment PCB data for San Jacinto). Texas Commission on Environmental Quality.

Krauskopf, K.B., 1979. *Introduction to Geochemistry*. McGraw Hill, New York.

Kushlan, J.A., 1978. Feeding ecology of wading birds. In: Wading Birds. A. Sprunt, J. Ogden, and S. Winckler (eds). National Audubon Society, New York, NY. pp. 249–296.

Litteer, D., 2009. Baytown Nature Center Bird Checklist. http://www.baytownnaturecenter.org/publications_information/bird_checklist.html. Accessed January 7, 2009.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder, 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19(1):81-97.

Lotrich, V.A. 1975. Summer home range and movements of *Fundulus heteroclitus* (Pisces: Cyprinodontidae) in a tidal creek. Ecology 56:191-198.

LSU, 2012. Biological Info: Black Drum. Accessed at http://www.seagrantfish.lsu.edu/biological/drum/blackdrum.htm. Accessed on February 6, 2012. Louisiana State University.

Macwhirter, B., P. Austin-Smith, Jr., and D. Kroodsma, 2002. Sanderling (*Calidris alba*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/653. Accessed October 26, 2011.

Martin, S., J. Duncan, D. Thiel, R. Peterson, and M. Lemke, 1989. Evaluation of the Effects of Dioxin-Contaminated Sludges on Eastern Bluebirds and Tree Swallows. Report prepared by Nekoosa Papers, Port Edwards, WI (as cited in Nosek et al. 1992a).

Mayer, F.L., P.M. Mehrle, and H.O. Sanders, 1977. Residue dynamics and biological effects of polychlorinated biphenyls in aquatic organisms. *Arch. Environ. Contam. Toxicol.* 5:501-511.Maxson, S.J., and L.W. Oring, 1980. Breeding season time and energy budgets of the polyandrous spotted sandpiper. *Behaviour* 74:200–263.

McCrady, J.J., and S.P. Maggard, 1993. Uptake and photodegradation of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin sorbed to grass foliage. *Environ. Sci. Technol.* 27:343-350.

McCrady, J.K., C. MacFarlane, and L.K. Gander, 1990. The transport and fate of 2,3,7,8-TCDD in soybean and corn. *Chemosphere* 21(3):359-376.

Meador, J.P., W.J. Adams, B.I. Escher, L.S. McCarty, A.E. McElroy, and K.G. Sappington, 2011. The tissue residue approach for toxicity assessment: Findings and critical reviews from a Society of Environmental Toxicology and Chemistry Pellston Workshop. *Integ. Environ. Assess. Manage.* 7(1)2-6.

Murata, M., 2003. Ecological risk assessment of dioxins for wild bird populations [dissertation]. Yokohama (JP): Yokohama National Univ (in Japanese).

Murata, M., S. Masunaga, and J. Nakanishi, 2003. Population-level ecological risk assessment of planar polychlorinated aromatic hydrocarbons in great cormorant (*Phalacrocorax carbo*) around Tokyo Bay. *Environ Toxicol Chem* 22:2508–2518.

Nacci, D.E., D. Champlin, L. Coiro, R. McKinney, and S. Jayaraman, 2002. Predicting the occurrence of genetic adaptation to dioxinlike compounds in populations of the estuarine fish (*Fundulus heteroclitus*). *Environmental Toxicology and Chemistry* 21(7):1525-1532.

Nagy, K.A., 2001. Food requirements of wild animals: Predictive equations for free-living mammals, reptiles, and birds. *Nutr. Abs. Rev. Ser. B: Livestock Feeds & Feeding* 71(10):21R–32R.

Naito, W., and M. Murata, 2007. Evaluation of population-level ecological risks of dioxin-like polychlorinated biphenyl exposure to fish-eating birds in Tokyo Bay and its vicinity. *Integrated Environmental Assessment and Management.* 3:68–78.

Naito, W., J. Jin, Y-S. Kang, M. Yamamuro, S. Masunaga, and J. Nakanishi, 2003. Dynamics of PCDDs/DFs and coplanar-PCBs in an aquatic food chain of Tokyo Bay. *Chemosphere* 53:347–362.

National Geographic, 2011. Alligator Snapping Turtle. Online publication accessed at http://animals.nationalgeographic.com/animals/reptiles/alligator-snapping-turtle/ on October 11, 2011.

Nebeker, A. V., and F. A. Puglisi, 1974. Effect of polychlorinated biphenyls (PCBs) on survival and reproduction of *Daphnia, Gammarus, and Tanytarsus*. *Trans. Amer. Fish. Soc.* 103: 722-728.

Nichols, M., J. Pruitt, D. Munsey, G. Good, B. Meyer, K. Urban, K. Francl and P. DiLaura, 1998. *Macrochelys temminckii* (On-line), Animal Diversity Web. Accessed October 11, 2011 http://animaldiversity.ummz.umich.edu/site/accounts/information/Macrochelys_temminckii.html.

Nicholson, A., B. Cornaby, G. Tiedemann, S. Krul, J. Hall, and J. Leech, 2012. Application of Site-Specific Sediment Exposure/Risk-Reduction Data for Remediation in an Industrial Lagoon. Battelle's Sixth International Conference on Remediation of Contaminated Sediments.

Niimi, A.J., 1996. PCBs in Aquatic Organisms. In: *Environmental Contaminants in Wildlife*. W.N. Beyer, G.H. Heinz, A.W. Redmon-Norwood (eds). Lewis Publishers, Boca Raton. pp. 117–151.

NOAA, 1995. San Jacinto River Bathymetric Survey H10619, June 13 – September 13, 1995. National Oceanic and Atmospheric Administration.

Nosek, J.A., S.R. Craven, J.R. Sullivan, J.R. Olsen, and R.E. Peterson, 1992a. Metabolism and Disposition of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin in Ring-Necked Pheasant Hens, Chicks, and Eggs. *Journal of Toxicology and Environmental Health* 35:153-164.

Nosek, J.A., S.R. Craven, J.R. Sullivan, S.S. Hurley, and R.E. Peterson, 1992b. Toxicity and Reproductive Effects of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin in Ring-Necked Pheasant Hens. *Journal of Toxicology and Environmental Health* 35:187-198.

Nosek, J.A., J.R. Sullivan, S.R. Craven, A.G. Gendron-Fitzpatrick, and R.E. Peterson, 1993. Embryotoxicity of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin in the Ring-Necked Pheasant. *Environmental Toxicology and Chemistry* 12:1215-1222.

Opperhuizen, A., and D.T.H.M. Sijm, 1990. Bioaccumulation and biotransformation of polychlorinated dibenzo-*p*-dioxins and dibenzofurans in fish. *Environ. Toxicol. Chem.* 9(2)175–186.

Osborn, M., P. Chai, A. Green, J. Lin, C. Loeffer, A. Morgan, P. Rubec, S. Spanyers, and A. Walton, 1992. Fisheries: In Status and Trends of Selected Living Resources in the Galveston Bay System, Eds. C. Loeffer and A. Walton. Galveston Bay National Estuary Program, Webster TX. GBNEP-19.

Portelli, M.J., and C.A. Bishop, 2000. *Ecotoxicolocgy of Amphibians and Reptiles.* Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.F. Powell, J.C. Restum, K.L. Stromborg, J.P. Giesy, and S.J. Bursian, 1997a. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126), 2,3,7,8-tetrachlorodibenzo-*p*-dioxinn (TCDD), or an extract derived from field-collected cormorant eggs injected into double-crested cormorant (*Phalacrocorax auritus*) eggs. *Environmental Toxicology and Chemistry* 16(7):1450-1455.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.P. Giesy, K.L. Stromborg, and S.J. Bursian, 1996. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) injected into the yolks of chicken (*Gallus domesticus*) eggs prior to incubation. *Arch Environ Contam Toxicol.* 31:404-409.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, M.E. Kelly, K.L. Stromborg, M.J. Melancon, S.D. Fitzgerald, and S.J. Bursian, 1998. Effects of 3,3',4,4',5-pentachlorobiphenyl and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin injected into the yolks of double-crested cormorant (*Phalacrocorax auritus)* eggs prior to incubation. *Environmental Toxicology and Chemistry* 17(10):2035-2040.

Quinney, T.E., 1982. Growth, diet, and mortality of nestling great blue herons. *Wilson Bull.* 94: 571-577.

R Core Development Team, 2008. *An Introduction to R: Notes on R, A Programming Environment for Data Analysis and Graphics* (electronic edition). Accessed at http://cran.r-project.org/doc/manuals/R-intro.pdf. Accessed on December 11, 2011.

RAIS, 2011. Risk Information System. http://rais.ornl.gov/. Accessed October 2011.

Riedle, J.D., D.B. Ligon, and K. Graves, 2008. Distribution and management of alligator snapping turtles, *Macrochelys temminckii*, in Kansas and Oklahoma. *Transactions of the Kansas Academy of Science* III(1/2):21-28.

Russell, R.W., F.A.P.C. Gobas, and G.D. Haffner, 1999. Maternal transfer and *in ovo* exposure of organochlorines in oviparous organisms: A model and field verification. *Environmental Science and Technology* 33(3):416-420.

Ryder, R.A., and D.E. Manry, 1994. White-faced Ibis (*Plegadis chihi*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/130.

Safe, S., 1990. Polychlorinated biphenyls (PCBs), dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: Environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). *Crit Rev Toxicol.* 21(1):51–88.

Sample, B.E., D.M. Opresko, and G.W.I. Suter, 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. ES/ER/TM-86/R3 Contract DE-AC05-84OR21400. Lockheed Martin Energy Systems, Inc., Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN. Appendix A.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W. Suter II, and T.L. Ashwood, 1998. Development and Validation of Bioaccumulation models for Earthworms. ES/ER/TM-220. Prepared for the US Department of Energy, Office of Environmental management. February 1998.

Sampson, J.R., R.A. Pastorok, and T.C. Ginn, 1996a. ER-L and ER-M values should not be used to assess contaminated sediments. *SETAC News*. September.

Sampson, J.R., R.A. Pastorok, and T.C. Ginn, 1996b. Response to MacDonald et al. (1996) concerning the derivation and use of ERL and ERM values. *SETAC News*. November.

Singh, A., R. Maichle, and S.E. Lee, 2006. On the Computation of a 95% Upper Confidence Limit of the Unknown Population Mean Based Upon Data Sets with Below Detection .Limit Observations. EPA/600/R-06/022. Lockheed Martin Environmental Services, Contract No. 68-W-04-005. Task Order No. 09. Las Vegas, NV.

Shields, M., 2012. Brown Pelican (*Pelecanus occidentalis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; http://bna.birds.cornell.edu/bna/species/609/articles/behavior?searchterm=Shields

Smithsonian (Smithsonian Marine Station at Fort Pierce), 2012. Black Drum. http://www.sms.si.edu/irlspec/pogoni_cromis.htm. Accessed on February 6, 2012.

Sparling, D.W., C.A. Bishop, and G. Linder (ed.), 2000b. *Ecotoxicology of Amphibians and Reptiles*. Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Staples, C.A., D.R. Peterson, T.F. Parkerton, and W.J. Adams, 1997. The environmental fate of phthalate esters: a literature review. *Chemosphere* 35: 667-749.

Steevens, J.A., M.R. Reiss, and A.V. Pawlisz, 2005. A methodology for deriving tissue residue benchmarks for aquatic biota: A case study for fish exposed to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and equivalents. *Integrated Environmental Assessment and Management* 1(2):142-151.

Stephens, R.D., M.X. Petreas, and D.G. Hayward, 1995. Biotransfer and bioaccumulation of dioxins and furans from soil: Chickens as a model for foraging animals. *Sci. Total Environ.* 175:253-273.

Suter, II, G.W. 1993. *Ecological Risk Assessment.* Lewis Publishers, Chelsea, MI.

Sutter, F.C., R.S. Waller, and T.D. McIlwain, 1982. Species Profiles: Black Drum. Accessed at http://www.nwrc.usgs.gov/wdb/pub/species_profiles/82_11-051.pdf. Accessed on February 6, 2012.

T&N Associates, 2002. Dose-Response Assessment from Published Research (1981-1997) of the Toxicity of 2,3,7,8-TCDD and Other Dioxin-like Compounds to Fish and Aquatic-

Associated Wildlife - Laboratory Studies. T&N Associates, Inc. Cincinnati, OH. September 2002.

TCEQ, 2006. Update to Guidance for Conducting Ecological Risk Assessments at Remediation Sites in Texas RG-263 (Revised). Remediation Division. January. http://www.tceq.state.tx.us/assets/public/remediation/eco/0106eragupdate.pdf

TCEQ and USEPA, 2006. Screening Site Assessment Report San Jacinto River Waste Pits, Channelview, Harris County, Texas. TXN000606611. Texas Commission on Environmental Quality and U.S. Environmental Protection Agency.

Telfair II, R.C., and M.L. Morrison, 2005. Neotropic Cormorant (*Phalacrocorax brasilianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/137

Teo, S.L.H., and K.W. Able, 2003. Habitat use and movement of the mummichog (*Fundulus heteroclitus*) in a restored marsh. Estuaries 26: 720-730.

Tietge, J.E., R.D. Johnson, K.M. Jensen, P.M. Cook, G.E. Elonen, and J.D. Fernandez, 1998. Reproductive toxicity and disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin in adult brook trout (*Salvelinus fontinalis*) following a dietary exposure. *Environ. Toxicol. Chem.* 17(12):2395–2407.

TPWD, 2009a. All Tackle Records for San Jacinto River. Available at http://www.tpwd.state.tx.us/fishboat/fish/action/alltackle.php?WB_code=1530. Texas Parks and Wildlife Department.

TPWD, 2009b. Wildlife Fact Sheets. Available at http://www.tpwd.state.tx.us/huntwild/wild/species. Texas Parks and Wildlife Department.

TPWD, 2010. Harris County ET List. Annotated County Lists of Rare Species. Available at: http://gis2.tpwd.state.tx.us/reportserver$gis_epasde_sql/pages/reportviewer.aspx?%2fr frepo+project2%2freport5&rs:command=render&county=harris. Texas Parks and Wildlife Department.

TPWD, 2012a. Species fact sheets: Black Drum. http://www.tpwd.state.tx.us/huntwild/wild/species/blackdrum. Accessed on February 7, 2012.

TPWD, 2012b. Bald Eagle (*Haliaeetus leuccocephalus).* Texas Parks and Wildlife, Electronic publication accessed at: http://www.tpwd.state.tx.us/publications/pwdpubs/media/pwd_bk_w7000_0013_bald_eagle.pdf. Accessed on February 6, 2012.

TPWD, 2012c. Rafinesque's Big-eared Bat (*Corynorhinus rafinesquii).* http://www.tpwd.state.tx.us/huntwild/wild/species/rafinesque. Accessed on February 26, 2012. Texas Parks and Wildlife Department.

TSHA, 2009. The San Jacinto River. Texas State Historical Association. Accessed at: http://www.tshaonline.org/handbook/online/articles/SS/rns9.html. Accessed on December 25, 2009.

UMMZ, 2011. Diversity Web. http://animaldiversity.ummz.umich.edu. Accessed September 26, 2011. University of Michigan, Museum of Zoology, Ann Arbor, MI.

University of Florida, 2005. Microcosm Evaluation of the Bioaccumulation of Organochlorine Pesticides from Soils in the North Shore Restoration Area at Lake Apopka. Final Report. Prepared for the Department of Water Resources, St. Johns River Water Management District, Palatka, FL. University of Florida, Gainesville, FL. July 2005.

University of Houston and Parsons, 2006. Total Maximum Daily Loads for Dioxins in the Houston Ship Channel. Contract No. 582-6-70860, Work Order No. 582-6-70860-02. Quarterly report No. 3. Prepared in cooperation with the Texas Commission on Environmental Quality and the U.S. Environmental Protection Agency. University of Houston and Parsons Water & Infrastructure. Available at: http://www.tceq.state.tx.us/assets/public/implementation/water/tmdl/26hscdioxin/26-all-data-compiled-q3-fy06.pdf.

University of Houston and Parsons, 2009. Total Maximum Daily Loads for PCBs in the Houston Ship Channel. Contract No. 582-6-60860. Work Order No. 528-6-60860-19. Draft Final Report. Prepared in cooperation with the Texas Commission on

Environmental Quality and the U.S. Environmental Protection Agency. University of Houston and Parsons Water & Infrastructure.

University of Texas, 1999. Herps of Texas: Frogs and Toads. January 1999. Available at: http://www.zo.utexas.edu/research/txherps/frogs.

URS, 2010. Data Usability Summary. Surface Water and Sediment Samples. San Jacinto River Waste Pits Superfund Site, Channelview, Harris County, Texas. Prepared for Texas Commission on Environmental Quality, Austin, Texas. Project No: 25335373. URS Corporation, Houston, TX.

USDOE, 2012. Risk Assessment Information System. Available at: http://rais.ornl.gov/. U.S. Department of Energy, Office of Environmental Management, Oak Ridge Operations Office, Oak Ridge, TN.

Usenko, S., B. Brooks, E. Bruce, and S. Williams, 2009. Defining Biota-Sediment Accumulation Factors for the San Jacinto River Waste Pits, Texas Project Work Plan and QAQC Procedures. Center for Reservoir and Aquatic Systems Research and the Department of Environmental Science, Baylor University. September 2009.

USEPA, 1985. Health Assessment Document for Polychlorinated Dibenzo-*p*-Dioxins. EPA/600/3-84/014F. Cincinnati, OH: Center for Environmental Research Information.

USEPA, 1986. Quality Criteria for Water 1986. EPA 440/5-86-001. May 1, 1986. U.S. Environmental Protection Agency, Office of Water, Regulations and Standards, Washington, DC.

USEPA, 1988. Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1989. Interim Procedures for Estimating Risks Associated with Exposure to Mixtures of Chlorinated Dibenzo-*p*-dioxins and Dibenzofurans (CDDs and CDFs) and 1989 Update. EPA 625/3-89/016. NTIS PB90-145756. U.S. Environmental Protection Agency, Risk Assessment Forum. Washington, DC.

USEPA, 1993. Exposure Factors Handbook-Volume 1. General Factors. EPA/600/P-95/002Fa. U.S. Environmental Protection Agency, National Center for

Environmental Assessment, Office of Research and Development, Washington, DC, and Versar Inc., Exposure Assessment Division, Springfield, VA.

USEPA, 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. EPA 540-R-97-006. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 1999a. Guidance for Conducting Risk Assessments at Hazardous Waste Combustion Sites. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 1999b. Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities Peer Review Draft, November 1999. Appendix C: Media-to-Receptor Bioconcentration Factors (BCFs). Available at: http://www.epa.gov/osw/hazard/tsd/td/combust/ecorisk.htm. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 1999c. Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities. EPA530-D-99-001A. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2001. Appendix D, Advanced Modeling Approaches for Characterizing Variability and Uncertainty. Risk Assessment Guide for Superfund Volume III - Part A: Process for Conducting Probabilistic Risk Assessment. EPA 540-R-02-002. Office of Emergency and Remedial Response, U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2003a. Attachment 1-3: Guidance for Developing Ecological Soil Screening Levels (Eco-SSLs): Evaluation of Dermal Contact and Inhalation Exposure Pathways for the Purpose of Setting Eco-SSLs. OSWER Directive 92857-55 November 2003

USEPA, 2003b. Generic Ecological Assessment Endpoints (GEAEs) for Ecological Risk Assessment. EPA/630/P-02/004F. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC. October.

USEPA, 2005a. Guidance for Developing Ecological Soil Screening Levels. OWSER Directive 9285.7-55. Revised February 2005. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2005b. Ecological Soil Screening Levels for Cadmium. Interim final. OSWER Directive 9285.7-65. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005c. Ecological Soil Screening Levels for Lead. Interim final. OSWER Directive 9285.7-70. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005d. Ecological Soil Screening Levels for Vanadium. Interim Final. OSWER Directive 9285.7-75. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007b. ProUCL version 4.0 technical guide. U.S. Environmental Protection Agency. U.S. EPA/600/R-07/041. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2007c. Guidance for Developing Ecological Soil Screening Levels. Attachment 4-1: Exposure Factors and Bioaccumulation Models for Derivation of Wildlife Eco-SSLs. Table 4a. http://www.epa.gov/ecotox/ecossl/SOPs.htm. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2007d. Ecological Soil Screening Levels for Copper. Interim final. OSWER Directive 9285.7-68. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007e. Ecological Soil Screening Levels for Nickel. Interim final. OSWER Directive 9285.7-76. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007f. Ecological Soil Screening Levels for Zinc. June 2007. OSWER Directive 9285.7-73 U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2008. Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. EPA 100/R-08/004. June 2008. U.S. Environmental Protection Agency, Office of the Science Advisor, Risk Assessment Forum, Washington, DC.

USEPA, 2009. Unilateral Administrative Order for Remedial Investigation/Feasibility Study. U.S. EPA Region 6 CERCLA Docket No. 06-03-10. In the matter of: San Jacinto River Waste Pits Superfund Site Pasadena, Texas. International Paper Company, Inc. & McGinnes Industrial Management Corporation, respondents.

USEPA, 2010a. Recommended toxicity equivalence factors (TEFs) for human health risk assessments of 2,3,7,8-tetrachlorodibenzo-p-dioxin and dioxin-like compounds. EPA/100-R-10/005. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC. December.

USEPA, 2010b. Final Report, Bioavailability of Dioxins and Dioxin-Like Compounds in Soil; prepared for U.S. Environmental Protection Agency, Office of Superfund Remediation and Technology Innovation, Environmental Response Team – West, Las Vegas, NV. Prepared by SRC, Inc., Chemical, Biological and Environmental Center, N. Syracuse, NY.

USEPA, 2012. Ecological Soil Screening Levels. Accessed at: http://www.epa.gov/ecotox/ecossl/. Accessed on March 8, 2012. U.S. Environmental Protection Agency, Washington, DC.

USFWS, 1985. Species Profiles: Life Histories and Environmental Requirements of Coastal Fishes and Invertebrates: Common Rangia. *U.S. Fish and Wildl. Biol. Rep* 82(11). U.S. Army Corps of Engineers, TR EL-82-4. U.S. Fish and Wildlife Service.

USFWS, 2008. Plants of Trinity River National Wildlife Refuge. Available at: http://www.fws.gov/southwest/refuges/texas/trinityriver/Docs/TrinityRiver85x11Plants2008.pdf. Downloaded on January 7, 2009. U.S. Fish and Wildlife Service.

USFWS, 2009a. Fish of Trinity River National Wildlife Refuge. Available at http://www.USFWS.gov/southwest/refuges/texas/trinityriver/Docs/TRFish2009.pdf. U.S. Fish and Wildlife Service.

USFWS. 2009b. Fish of Anahuac National Wildlife Refuge. Accessed at http://www.USFWS.gov/southwest/refuges/texas/anahuac/fishingandboating.html Accessed on January 4, 2010. U.S. Fish and Wildlife Service.

USFWS, 2009c. Amphibians and Reptiles of Trinity River National Wildlife Refuge. Available at:

http://www.fws.gov/southwest/refuges/texas/trinityriver/docs/trnwrherps2009.pdf. U.S. Fish and Wildlife Service.

USGS, 2007. Glossary of Terms. http://water.usgs.gov/nawqa/glos.html. Last updated January 10, 2007. Accessed on February 4, 2012. U.S. Geological Survey.

USGS, 2009. Nonindigineous aquatic species. Accessed at http://nas.er.usgs.gov/queries/FactSheet.asp?speciesID=92. Accessed on January 7, 2010. U.S. Geological Survey.

Van den Berg, M., L. Bimbaum, A.T. Bosveld, B. Brumstrom, P. Cook, M. Feeley, J.P. Giesy, A. Hanberg, R. Hasegawa, S.W. Kennedy, T. Kubiak, J.C. Larsen, F.X. va Leeuwen, A.K. Liem, C. Nolt, R.E. Peterson, L. Poellinger, S. Safe, D. Schrenk, D. Tillitt, M. Tysklind, M. Younes, F. Waern, and T. Zacharewski, 1998. Toxic Equivalency Factors (TEFs) for PCBs, PCDDs, PCDFs for Humans and Wildlife. *Environmental Health Perspectives* 106:775-792.

Van den Berg, M., L.S. Birnbaum, and M. Denison, 2006. The 2005 World Health Organization Reevaluation of Human and Mammalian Toxic Equivalency Factors for Dioxins and Dioxin-Like Compounds. *Toxicol. Sci.* 93(2):223-241.

Wan, Y., J. Hu, M. Yang, L. An, W. An, X. Jin, T. Hattori, and M. Itoh, 2005. Characterization of trophic transfer for polychlorinated dibenzo-*p*-dioxins, dibenzofurans, non- and mono-ortho polychlorinated biphenyls in the marine food web of Bohai Bay, North China. *Environ. Sci. Technol.* 36(8):2417–2425.

Weir, S.M., J.G. Suski, and C.J. Salice, 2010. Ecological risk of anthropogenic pollutants to reptiles: Evaluating assumptions of sensitivity and exposure. *Environmental Pollution* 158:3596-3606.

WHO, 2001. Vanadium Pentoxide and Other Inorganic Vanadium Compounds. Concise International Chemical Assessment Document 29. ISBN 92 4 153029 4.

Wild, S.R., and K.C. Jones, 1992. Organic chemicals entering agricultural soils in sewage sludges: screening for their potential to transfer to crop plants and livestock. *Sci. Total Environ.* 119:85–119.

Windward, 2007. Phase 2 Remedial Investigation Report Appendix A: Baseline Ecological Risk Assessment for the Lower Duwamish Waterway Group. Seattle, WA. http://www.ldwg.org/assets/era/final_era_7_31_07.pdf. Accessed on February 6, 2012.

Wintermyer, M.L., and K.R. Cooper, 2007. The Development of an Aquatic Bivalve Model: Evaluating the Toxic Effects on Gametogenesis following 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin (2,3,7,8-TCDD) Exposure in the Eastern Oyster (*Crassostrea virginica*). *Aquatic Toxicology* (Amsterdam) 81(1):10-26.

Wolfe, J.L., 1982. *Oryzomys palustris*. Mammalian Species No. 176, pp 1-5. *American Society of Mammalogists*. 18 June 1982.

Woodford, J.E., W.H. Karasov, M.W. Meyer, and L. Chambers, 1998. Impact of 2,3,7,8-TCDD Exposure on Survival, Growth, and Behavior of Ospreys Breeding in Wisconsin, USA. *Environmental Toxicology and Chemistry* 17(7):1323-1331.

Young, H., 2010. Personal Communication (email exchange between D. Rudnick, Integral Consulting Inc. and H. Young, NOAA, discussing molluscs likely to be found in upper Galveston Bay). January 13, 2010.

Zwiernik, M.J., K.J. Beckett, S. Bursian, D.P. Kay, R.R. Holem, J.N. Moore, B. Yamini, and J.P. Giesy, 2009. Chronic effects of polychlorinated dibenzofurans on mink in laboratory and field environments. *Integr. Environ. Assess. Management* 5(2):291-301.

TABLES

Table 3-1
Chemicals of Potential Ecological Concern

Chemical	Receptors North of I-10 and Aquatic Environment	
	Benthic Invertebrates	Fish and Wildlife
Dioxins/Furans		
Dioxins and Furans	X	X
Polychlorinated Biphenyls		
Polychlorinated Biphenyls		X
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	X	X
Carbazole	X	
Phenol	X	
Metals		
Aluminum	X	
Barium	X	
Cadmium		X
Cobalt	X	
Copper	X	X
Lead	X	
Manganese	X	
Mercury	X	X
Nickel		X
Thallium	X	
Vanadium	X	
Zinc	X	X

Notes

$COPC_E$ = chemical of potential ecological concern

Table 3-2
Toxicity Equivalency Factors for Dioxins and Furans and Dioxin-Like PCBs

Compound	TEF-M (WHO 2005)[a]	TEF-Fish (WHO 1998)	TEF-Bird (WHO 1998)
Chlorinated Dibenzo-*p*-dioxins			
2,3,7,8-TCDD	1	1	1
1,2,3,7,8-PeCDD	1	1	1
1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
1,2,3,4,6,7,8-HpCDD	0.01	0.001	0.001
OCDD	0.0003	0.0001	0.0001
Chlorinated Dibenzofurans			
2,3,7,8-TCDF	0.1	0.05	1
1,2,3,7,8-PeCDF	0.03	0.05	0.1
2,3,4,7,8-PeCDF	0.3	0.5	1
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
OCDF	0.0003	0.0001	0.0001
Non-ortho Substituted PCBs			
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.0001	0.0001	0.05
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.0003	0.0005	0.1
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	0.005	0.1
3,3',4,4',5,5'-Hexachlorobiphenyl (PCB 169)	0.03	0.00005	0.001
Mono-ortho Substituted PCBs			
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.00003	0.000005	0.0001
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.00003	0.000005	0.0001
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00003	0.000005	0.00001
2',3,4,4',5-Pentachlorobiphenyl (PCB 123)	0.00003	0.000005	0.00001
2,3,3',4,4',5-Hexachlorobiphenyl (PCB 156)	0.00003	0.000005	0.0001
2,3,3',4,4',5'-Hexachlorobiphenyl (PCB 157)	0.00003	0.000005	0.0001
2,3',4,4',5,5'-Hexachlorobiphenyl (PCB 167)	0.00003	0.000005	0.00001
2,3,3',4,4',5,5'-Heptachlorobiphenyl (PCB 189)	0.00003	0.000005	0.00001

Sources

WHO (1998) corresponds to Van den Berg et al. (1997)

WHO (2005) corresponds to Van den Berg et al. (2006)

Notes

PCB = polychlorinated biphenyl
TEF-M = mammalian toxicity equivalency factor
a - Endorsed by USEPA (2010a)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Amphibians				
Gulf Coast toad	*Bufo valliceps valliceps*	From coastal prairies and barrier beaches along the Gulf of Mexico to roadside and irrigation ditches to urban/suburban sewers and backyard gardens		University of Texas (1999)
Southern leopard frog	*Rana sphenocephala utricularia*	All types of shallow freshwater habitats, including temporary pools, cypress ponds, ponds, lakes, ditches, irrigation canals, and stream and river edges; will inhabit slightly brackish coastal wetland		USFWS (2009c); TPWD (2009b)
Reptiles				
American alligator	*Alligator mississippiensis*	Alligators are found in or near water. They are common in swamps, rivers, bayous, and marshes. While typically found in fresh-water, they can tolerate brackish water as well.		USFWS (2009c)
Western cottonmouth	*Agkistrodon piscivorus leucostor*	Western cottonmouths prefer lowland swamps, lakes, rivers, sloughs, irrigation ditches, rice fields and salt marshes, but are not confined to living in moist habitats		USFWS (2009c)
Gulf Salt Marsh snake	*Nerodia clarkii*	Just as the name indicates, gulf salt marsh snakes prefer brackish and saltwater estuaries, salt marshes and tidal mud flats.	R	USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Texas garter snake	*Thamnophis sirtalis annectens*	Wet or moist microhabitats are conducive to the species occurrence, but is not necessarily restricted to them; hibernates underground or in or under surface cover; breeds March-August	R	TPWD (2010)
Timber rattlesnake	*Crotalus horridus*	Swamps, floodplains, upland pine and deciduous woodlands, riparian zones, abandoned farmland; limestone bluffs, sandy soil or black clay; prefers dense ground cover, i.e., grapevines or palmetto	T	TPWD (2010)
Smooth green snake	*Liochlorophis vernalis*	Gulf Coastal Plain; mesic coastal shortgrass prairie vegetation; prefers dense vegetation	T	TPWD (2010)
Common snapping turtle	*Chelydra serpentina*	The snapping turtle can be found in waters ranging from slow moving rivers to stagnant ponds.		USFWS (2009c)
Alligator snapping turtle	*Macrochelys temminickii*	Alligator snapping turtles generally live in the deepest water within their habitat: large rivers, canals, lakes, swamps, and rivers.	T	USFWS (2009c)
Western chicken turtle	*Deirochelys reticularia maria*	Chicken turtles are semi-aquatic turtles, found both in water and on land. They prefer water with dense vegetation and soft substrate.		USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Eastern river cooter	*Psuedemysconcinna metteri*	The river cooter is primarily a river turtle, but can be found in ditches and saltwater areas near river mouths. Rivers with slow to moderate currents, abundant aquatic vegetation, and rocky bottoms are preferred. Other less frequently used habitats include lakes, ponds, deep springs, floodplain river pools, and swamps.		USFWS (2009c)
Common musk turtle	*Sternotherus odoratus*	The habitat of the common musk turtle includes any kind of permanent body of water, like shallow streams, ponds, rivers, or clear water lakes, and it is rare to find the turtle elsewhere.		USFWS (2009c)
Red-eared sldier	*Trachemys scripta elegans*	The red-eared slider enjoy large areas where they are free to swim. These turtles also require a basking area, where they can completely leave the water and enjoy the light provided for them.		USFWS (2009c)
Texas spiny softshell turtles	*Trionyx spiniferus emoryi*	Soft-shelled turtles are almost entirely aquatic powerful swimmers, fond of basking and rarely venture far from aquatic margins.		USFWS (2009c)

Table 3-3
Reptiles and Amphibians That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations[a]	Federal and/or State Listing	Source of Information
Diamondback terrapin	*Malaclemys terrapin littoralis*	Diamondback terrapins prefer brackish or salt water. They are the only turtle found in estuaries, tidal creeks, and saltwater marshes where the salinity comes close to that of the ocean.	R	TPWD (2009b)

Notes

a - Additional habitat Information accessed at www.amphibiaweb.org and http://animaldiversity.ummz.umich.edu/site/index.html

Federal or State Listing

LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Benthic			
Omnivores			
Pinfish	*Lagodon rhomboides*	Commonly found on vegetated bottoms, occasionally over rocky bottoms and in mangrove areas. Enters brackish water and even freshwaters. Feeds mainly on small animals, especially crustaceans, but also takes mollusks, worms and occasionally small fishes that are associated with the grassy habitat.	Osborn et al. (1992)
Atlantic croaker	*Micropogonias undulatus*	Occurs usually over mud and sandy mud bottoms in coastal waters and in estuaries where the nursery and feeding grounds are located. Feeds mainly on worms, crustaceans and fishes.	Osborn et al. (1992)
Hardhead catfish	*Ariopsis felis*	Inhabits continental waters and enters estuaries. Found in turbid waters over muddy bottoms. Commonly captured from catwalks, bridges and piers, particularly in passes and inland waterways. It has a varied diet including detritus, invertebrates, and fish.	Crocker and Young (1990)
Carnivores			
Blue catfish	*Ictalurus furcatus*	Diet is variable, tends to eat fish earlier in life. Also uses invertebrates Inhabits deep water of impoundments, main channels, and backwaters of medium to large rivers, over mud, sand and gravel.	TPWD (2009a)
Channel catfish	*Ictalurus punctatus*	Estuaries, lagoons, brackish seas, rivers, streams, lakes and ponds. Feed primarily on small fish, crustaceans (e.g., crayfish), clams and snails; also feed on aquatic insects and small mammals.	TPWD (2009a)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Southern flounder	*Paralichthys lethostigma*	Found mostly over mud bottoms in estuaries and coastal waters to about 40 m depth. A cryptic species; tolerates low salinities; occurs frequently in brackish bays and estuaries, even on occasion in fresh water. This species moves to deeper water in winter, but is still easily accessible. Feeds chiefly on fishes, also on crabs and shrimps. Juveniles take mainly small bottom-living invertebrates	Osborn et al. (1992)
Bowfin [b]	*Amia calva*	Found in swampy, vegetated lakes and rivers. An air-breather that can withstand high temperatures, which enables it to survive in stagnant areas and is even known to aestivate; lethal temperature is 35.2°C. A voracious and opportunistic feeder, it uses scent as much as site and subsists on fish, frogs, crayfish, insects, and shrimps.	TPWD (2009b)
Pelagic			
Omnivore			
Grass carp	*Ctenopharyngodon idella*	Occurs in lakes, ponds, pools and backwaters of large rivers, preferring large, slow-flowing or standing water bodies with vegetation. Tolerant of a wide range of temperatures from 0° to 38°C, and salinities to as much as 10 ppt and oxygen levels down to 0.5 ppm. Feeds on higher aquatic plants and submerged grasses; takes also detritus, insects and other invertebrates.	USFWS (2009a)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Invertivore			
Gulf killifish	*Fundulus grandis*	Small fish species common in estuaries and rivers of the Gulf Coast. They do not migrate, remaining in the same location for their entire life. They eat various small invertebrates. Tolerates a wide range of salinities, from freshwater to estuarine.	Hassan-Williams et al. (2007)
Carnivore			
Dollar sunfish [b]	*Lepomis marginatus*	Inhabits sand-bottomed and mud-bottomed, usually brushy, pools of creeks and small to medium rivers; and also swamps. Feeds on midge larvae and microcrustacean.	TPWD (2009a)
Red drum	*Sciaenops ocellatus*	Occurs usually over sand and sandy mud bottoms in coastal waters and estuaries. Abundant in surf zone. Feeds mainly on crustaceans, mollusks and fishes.	TPWD (2009a)
Black drum	*Pogonias cromis*	Usually found over sand and sandy mud bottoms in coastal waters, especially in areas with large river runoffs. Juveniles often enter estuaries. Primarily a benthic feeder, mainly on crustaceans, molluscs and fishes.	Osborn et al. (1992)
Spotted seatrout	*Cynoscion nebulosus*	Inhabits river estuaries and shallow coastal marine waters over sand bottoms, often associated with seagrass beds. Also occurs in salt marshes and tidal pools of high salinity. Feeds mainly on crustaceans and fishes.	Osborn et al. (1992)

Table 3-4
Fish That May Be Found in the Vicinity of the San Jacinto River Waste Pit Site

Common Name	Scientific Name	Habitat Associations and Diet[a]	Source of Information
Bay anchovy	*Anchoa mitchilli*	More commonly found in shallow tidal areas with muddy bottoms and brackish waters, tolerates a wide range of salinities (virtually fresh to fully saline or hypersaline). Feeds mostly on Mysis and copepods, also small fishes, gastropods, and isopods.	Osborn et al. (1992)

Notes

a - Additional habitat association information from www.fishbase.org

b - Found rarely in estuaries

Table 3-5
Invertebrates That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Source of Information
Blue crab	*Callinectus sapidus*	Blue crabs are benthic in every type of habitat from the saltiest water of the gulf to the almost fresh water of the back bays and estuaries, from the low tide line to waters 120 ft (36 m) deep. It is considered a scavenger, eating dead or dying organisms, but will also take live prey.	Crocker and Young (1990)
Oyster	*Crassostrea virginica*	Eastern oysters are abundant in shallow saltwater bays, lagoons and estuaries, in water 8 to 25 feet (2.5 to 7.5 m) deep and between 28 and 90 degrees F. Have been collected in the vicinity of the Site.	Crocker and Young (1990), Broach (2010)
Stone crab	*Menippe mercenaria*	Stone crabs prefer bottoms of bays, oyster reefs and rock jetties where they can burrow or find refuge from predators. Juveniles do not usually dig burrows, but instead hide among rocks or in seagrass beds.	TPWD (2009b)
Hermit crab	*Clibanarius vittatus*	Benthic scavengers found in the intertidal.	GBIC (2009)
Fiddler crab	*Uca longisignalis*	Fiddler crabs are most often found in soft sand or mud near or around the edges of shallow salt marshes.	TPWD (2009b)
Asian clam	*Corbicula fluminea*	Sand and clay, salinities up to 13 ppt.	USGS (2009)
Common rangia	*Rangia cuneata*	Low salinity estuaries, <19 ppt, most found in 5 - 15 ppt. Found in sandy, muddy, and vegetated areas. Species has been collected from the vicinity of the Site.	USFWS (1983), Broach (2010)
Brown rangia	*Rangia flexuosa*	Typically found in the intertidal zone at the water's edge. Species has been collected from the vicinity of the Site.	Broach (2010)
Dark false mussel	*Mytilopsis leucophaeata*	Typically found in brackish waters.	Broach (2010)
Dwarf surf clam	*Mulinia lateralis*	The dwarf surf clam is normally found in the soft strata in benthic communities.	Broach (2010)
Surf clam	*Macoma mitchelli*		Young (2010)

Table 3-5
Invertebrates That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Source of Information
Hooked mussel	*Ischadium recurvum*	Typically found in the intertidal zone at the water's edge. Species has been collected from the vicinity of the Site.	Culbertson (2010)
Southern quahog	*Mercenaria texana*		Culbertson (2010)
Grass shrimp	*Palaemonetes pugio*	A small shrimp species common to the estuaries of the Gulf Coast. Short life span (6-12 months). Limited commercial, recreational, or consumptive value for humans, but is a food source for many other species. Inhabits low salinity areas with grassy shorelines.	GBIC (2009)

Table 3-6
Aquatic and Wetland Plants That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Federal and/or State Listing	Source of Information
Water-milfoil	*Myriophyllum pinnatum*		USFWS (2008)
Threeflower broomweed	*Thurovia triflora*	R	TPWD (2010)
Eastern woodland sedge	*Carex blanda Dewey*		USFWS (2008)
Thinfruit sedge	*Carex flaccosperma Dewey*		USFWS (2008)
Frank's sedge	*Carex frankii Kunth*		USFWS (2008)
Shoreline sedge	*Carex hyalinolepis Steud.*		USFWS (2008)
Greater bladder sedge	*Carex intumescens Rudge*		USFWS (2008)
Cypress swamp sedge	*Carex joorii L.H. Bailey*		USFWS (2008)
Blunt broom sedge	*Carex tribuloides Wahlenb.*		USFWS (2008)
Fox sedge	*Carex vulpinoidea Michx.*		USFWS (2008)
Common spikerush	*Eleocharis palustris*		USFWS (2008)
Blunt spikerush	*Eleocharis obtusa*		USFWS (2008)
Shortbristle horned beaksedge	*Rhynchospora corniculata*		USFWS (2008)
Scouring-rush	*Equisetum hyemale*		USFWS (2008)
Carolina foxtail	*Alopecurus carolinianus*		USFWS (2008)
Giant cutgrass	*Zizaniopsis miliacea*		USFWS (2008)
Jungle rice	*Echinochloa colona*		USFWS (2008)
Field paspalum	*Paspalum laeve Michx.*		USFWS (2008)
Southern canary grass	*Phalaris caroliniana*		USFWS (2008)
Cattail	*Typha latifola*		USFWS (2008)
Tapertip rush	*Juncus acuminatus*		USFWS (2008)
Forked rush	*Juncus dichotomus*		USFWS (2008)
Common rush	*Juncus effusus*		USFWS (2008)
Inland rush	*Juncus interior*		USFWS (2008)
Grassleaf rush	*Juncus marginatus*		USFWS (2008)
Path rush	*Juncus tenuis*		USFWS (2008)
Flat rush	*Juncus validus*		USFWS (2008)
Common duckmeat	*Spirodela polyrrhiza*		USFWS (2008)
Duckweed	*Lemna aequinoctialis*		USFWS (2008)
Water-meal	*Wolffia brasiliensis*		USFWS (2008)
Water-meal	*Wolffia columbiana*		USFWS (2008)
Marsh purslane	*Ludwigia palustris*		USFWS (2008)
Hairy water primrose	*Ludwigia grandiflora*		USFWS (2008)
Texas prairie dawn	*Hymenoxys texana*	LE, E	TPWD (2010)
Water lettuce	*Pistia stratiotes*		Gonzalez et al. (2006)
Common water hyacinth	*Eichornnia crassipes*		Gonzalez et al. (2006)

Federal or State Listing

LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare (State; this does not indicate a regulatory listing status)

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Omnivore			
Gadwall	*Anas strepera*	Dabbling duck, primarily herbivorous but feeds on invertebrates during breeding season. Wetlands, ponds, marshes and lakes with heavily vegetated margins.	
Green winged teal	*Anas crecca*	Opportunistic feeder; seeds of aquatic vegetation, also invertebrates. Found in shallow ponds and marshes with abundant vegetation, tidal creeks and mudflats.	
Northern pintail	*Anas acuta*	Nests in open country with shallow, seasonal wetlands and low vegetation. Winters in wide variety of shallow inland freshwater and intertidal habitats.	
Blue-winged teal	*Anas discors*	Variable diet, including aquatic invertebrates, seeds and algae. Shallow ponds and wetlands.	
Mallard	*Anas platyrhynchos*	From large marshes to small river bends and bays; found in a wide variety of habitats. Variety of vegetation, increased feeding on invertebrates during breeding season.	
Black-bellied whistling duck	*Dendrocygna autumnalis*	Primarily feeds on plant material, but also consumes insects and molluscs. Breeds in coastal Texas. Primarily breeds in shallow freshwater ponds and lakes.	
Northern shoveler	*Anas clypeata*	Freshwater marshes, tidal bays in winter	
Lesser scaup	*Aythya, affinis*	Salt marshes, estuaries and lakes	
Ring-billed gull	*Larus delawarensis*	Breeds on islands in inland lakes, in winter along seacoasts	
Laughing gull	*Larus atricilla*	Nests in marshes, on beaches, and on islands along coast Found along coasts, in estuaries, bays, and inland lakes. Feeds along the ocean, on rivers, at landfills, and in urban parks.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Gull-billed tern	*Sterna nilotica*	Breeds on gravelly or sandy beaches. Winters in salt marshes, estuaries, lagoons and plowed fields, less frequently along rivers, around lakes and in fresh-water marshes.	
Roseate spoonbill	*Platalea ajaja*	Marsh habitat. Omnivore with a wide diet inluding plants, invertebrates and fish.	
Killdeer	*Charadrius viciferous*	Fields, coastal fields, beaches, lawns. Insects make up the majority of the killdeer's diet, but they will also eat berries and crustaceans.	
Invertivore			
Pied-billed grebe	*Podilymbus podiceps*	Breeds on seasonal or permanent ponds with dense stands of emergent vegetation, bays and sloughs. Uses most types of wetlands in winter.	
Least sandpiper	*Calidris minutilla*	Breeds in mossy or wet grassy tundra, occasionally in drier areas with scattered scrubby bushes. Migrates and winters in wet meadows, mudflats, flooded fields, shores of pools and lakes, and, less frequently, sandy beaches.	
Mottled duck	*Anas fulvigula*	Freshwater wetlands, ditches, wet prairies, and seasonally flooded marshes.	
Black-necked Stilt	*Himantopus mexicanus*	Shallow fresh and saltwater wetlands, including salt ponds, rice fields, shallow lagoons, and mangrove swamps	
Greater yellowlegs	*Tringa melanoleuca*	Breeds in muskeg, wet bogs with small wooded islands, and forests (usually coniferous) with abundant clearings. Winters in wide variety of shallow fresh and saltwater habitats.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Lesser yellowlegs	*Tringa flavipes*	Breeds in open boreal forest with scattered shallow wetlands. Winters in wide variety of shallow fresh and saltwater habitats.	
Spotted sandpiper	*Actitis macularia*	Breeds in a variety of habitats, such as shoreline, sagebrush, grassland, forest, lawn, or park. Winters wherever water is present. The spotted sandpiper is a shorebird that obtains much of its diet by probing or "mining" soft sediments along shorelines. Spotted sandpipers feed on a wide variety of benthic invertebrates and appear to be relatively common winter residents in coastal Texas.	
Western sandpiper	*Calidris mauri*	Breeds in coastal sedge-dwarf tundra. Migrates and winters along mudflats, beaches, shores or lakes and ponds, and flooded fields.	
White-faced ibis	*Plegadus chihi*	Primarily freshwater wetlands, but can also be found in estuarine habitats. Feeds on crustaceans, earthworms and insects	T
Carnivore			
Brown pelican	*Pelacanus occidentalis*	Oceans, inshore waters; stands on pilings or rocks	E
Double crested cormorant	*Phalacrocorax auritus*	Found in diverse aquatic habitats, such as ponds, lakes, rivers, lagoons, estuaries, and open coastline; more widespread in winter	
Neotropic cormorant	*Phalacrocorax brasilianus*	Various wetlands, including fresh, brackish, and saltwater habitats. Nests and roosts mostly in trees, but also on cliffs and human-made structures. Feeds primarily on fish <8cm in length.	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Great blue heron	*Ardea herodias*	Wetlands where tall trees, rock ledges or extensive reeds provide a safe site for the heronry. Feeds on fish but also crustaceans, amphibians.	
Great egret	*Casmerodius albus*	Marshes where deeper water is edged with low , vegeatated banks. Nesting colonies may be in reeds or cattails, but more commonly in trees.	
Tricolored heron	*Egretta tricolor*	Breeds primarily in coastal habitats; feeds mainly on small fishes.	
Little blue heron	*Egretta caerulea*	Swamps, estuaries, rivers, ponds, and lakes	
Snowy egret	*Egretta thula*	Near freshwater lakes or estuaries	
Cattle egret	*Bubucus ibis*	Extensive marshes, wooded marshes	
Green heron	*Butorides virescens*	Breeds in swampy thickets. Forages in swamps, along creeks and streams, in marshes, ponds, lake edges, and pastures. Winters mostly in coastal areas, especially mangrove swamps.	
Black-crowned night-heron	*Nycticorax nycticorax*	Various wetland habitats, including salt, brackish, and freshwater marshes, swamps, streams, lakes, and agricultural fields.	
Yellow-crowned night-heron	*Nyctanassa violacea*	Marsh	
White ibis	*Eudocimus albus*	Large marshes	
Red-breasted merganser	*Mergus serrator*	Lakes rivers, winters on saly water	
Osprey	*Pandion haliaetus*	Coasts and inland lakes and rivers	
Forster's tern	*Sterna forsteri*	Breeds in marshes, generally with lots of open water and large stands of island-like vegetation.Winters in marshes, coastal beaches, lakes, and rivers.	
Least tern	*Sterna antillarum*	Beaches, bordering, shallow water along rivers, lakes, or coasts	

Table 3-7
Aquatic-Dependent Birds That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations and Diet	Federal and/or State Listing
Belted kingfisher	*Megaceryle alcyon*	Breeds along streams, rivers, lakes, and estuaries with banks for nest holes. Winters along coast, streams, and lakes.	
Bald eagle	*Haliaeetus leucocephalus*	Coasts and inland lakes and rivers	BGEPA
Reddish egret	*Egretta rufescens*	Marsh habitat	

Notes

Birds are all listed on the bird checklist of the Baytown Nature Center (2006).
Additional habitat information from Cornell Lab of Ornithology's 2009 Bird Search. Accessed at
http://www.allaboutbirds.org/NetCommunity/Page.aspx?pid=1189 Accessed on December 30 2009, and from Birds of North America Online,

Federal or State Listing

BGEPA = Bald and Golden Eagle Protection Act
LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-8
Aquatic-Dependent Mammals That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Federal and/or State Listing	Source of Information
Herbivore				
Marsh rice rat	*Oryzomys palustris*	Marsh rice rats are semi-aquatic rodents that eats aquatic plants, and some invertebrates such as crabs and snails. This animal nests in cattatils and bulrushes, and is prey to hawks and owls.		eNature.com
Nutria	*Myocastor coypus*	Nutria are an invasive species that spend most of their time in or near the water. Favored foods for nutria include rushes, reeds, cattails, arrowhead, square-stem spike rush and sawgrass.		USFWS (2009)
American beaver	*Castor canadensis*	Herbivore found in ponds, lakes, or large streams.		USFWS (2009)
Omnivore				USFWS (2009)
Virginia opossum	*Didelphis virginiana*	Opossums are omnivorous, primarily woodland creatures, but are also frequently found in prairies, marshes, and farmlands. Although they prefer to live in hollow trees and logs, opossums will also shelter in woodpiles, rock piles, crevices in cliffs, under buildings, in attics, and in abandoned underground burrows dug by other animals.		USFWS (2009)
Northern raccoon	*Procyon lotor*	Raccoons prefer brushy or wooded areas near streams, lakes or swamps, although they can live close to developed areas if sufficient food, water and cover are provided. Though they prefer woodlands, raccoons can live practically anywhere and have adapted well to human habitats.		USFWS (2009)

Table 3-8
Aquatic-Dependent Mammals That May Be Found in the Vicinity of the San Jacinto River Waste Pits Site

Common Name	Scientific Name	Habitat Associations	Federal and/or State Listing	Source of Information
Muskrat	*Ondatra zibethicus*	Muskrats primarily inhabit wetlands, areas in or near salt and fresh-water marshlands, rivers, lakes, or ponds.		USFWS (2009)
Carnivore				USFWS (2009)
River otter	*Lutra canadensis*	River otters prefer to live near bodies of water such as lakes, large rivers, and streams. Along the Texas Gulf Coast region, otters also live in marshes, bayous, and brackish inlets.		USFWS (2009)
Insectivore				
Rafinesque's big-eared bat	*Corynorhinus rafinesquii*	Roosts in cavity trees of bottomland hardwoods, concrete culverts, and abandoned man-made structures	T	TPWD (2010)
Southeastern myotis	*Myotis austroriparius*	Roosts in cavity trees of bottomland hardwoods, concrete culverts, and abandoned man-made structures	R	TPWD (2010)

Federal or State Listing

LE/LT = Federally Listed Endangered/Threatened
E/T = State Endangered/Threatened
R = Rare

Table 3-9
Summary of Ecological Receptor Surrogates

Receptor Group	Receptor Surrogate	Feeding Guild	Potentially Present	Representative of One or More Feeding Guilds	High Site Fidelity/Residential	Sensitive or Potentially Highly Exposed	Life History Information Is Readily Available	Additional Considerations
Benthic macroinvertebrates								
	Benthic macroinvertebrate community	All	X	X	X	X	X	Close association with sediment; much of the toxicological literature addresses community level endpoints.
	Molluscs	Filter feeders	X	X	X	X[a]	X	Close association with sediment
Fish								
	Gulf killifish	Omnivore	X	X	X		X	Common prey for other fish and bird species
	Black drum	Benthic invertivore	X	X	X		X	Popular sport fish; limited range, limited interbay movement
	Southern flounder	Benthic piscivore	X	X	X[b]	X	X	Supports commercial and recreational fisheries
Reptiles								
	Alligator snapping turtle	Omnivore	X	X	X	X	X	Sensitive species (rare in estuaries)
Birds								
	Neotropic cormorant	Piscivore (diving)	X	X			X	
	Great blue heron	Piscivore (wading)	X	X			X	
	Spotted sandpiper	Invertivore (probing)	X	X		X	X	As a sediment-probing invertivore, expected to be closely associated with sediment exposure pathway
	Killdeer	Invertivore (terrestrial)	X	X	X		X	Feeds on invertebrate fauna closely associated with soils
Mammals								
	Marsh Rice Rat	Omnivore	X	X	X		X	Semi-aquatic, diet consists of aquatic and emergent plants, and invertebrates
	Raccoon	Omnivore	X	X			X	Representative of both aquatic and terrestrial omnivorous feeding guilds

Notes

a - Sensitive reproductive endpoint

b - Site fidelity is probably high except in winter, when this species moves into more saline waters to spawn.

Table 3-10
Summary of Assessment Endpoints and Risk Questions for the BERA

Receptor Class	Assessment Endpoint	Risk Questions
Benthic macroinvertebrates	Abundance and diversity of benthic macroinvertebrate communities	Are the concentrations of chemicals of potential concern ($COPC_E$s) in whole sediment from benthic habitats of the Site greater than threshold concentrations relating to the survival, growth, or reproduction of benthic invertebrates, or the productivity or viability of invertebrate populations or communities?
Bivalve molluscs	Stable or increasing populations of bivalves within the Site	Are concentrations of organic primary $COPC_E$s in tissue of field collected clams equal to or greater than concentrations considered threshold levels of reproductive effects in molluscs?
Fish	Stable or increasing populations of fish in the following feeding guilds:	Are the concentrations of $COPC_E$s in waters of the Site greater than threshold concentrations relating to the survival, growth, or reproduction of fish?
	- Benthic omnivore - Benthic invertivore - Benthic piscivore	Are the concentrations of inorganic $COPC_E$s (metals) in the diet of fish greater than threshold effect levels for survival, growth, or reproduction of fish?
		Are concentrations of organic $COPC_E$s in fish tissue from the Site greater than the concentrations of $COPC_E$s associated with effects on the survival, growth or reproduction of fish?
Reptiles	Stable or increasing populations of omnivorous reptiles	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth and reproduction of reptiles?
Birds	Stable or increasing populations of birds (that may be exposed to $COPC_E$s from the Site) in the following feeding guilds:	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth, and reproduction of birds? Is the estimated concentration of dioxins and furans, expressed as TEQs, in bird eggs greater than threshold concentrations for reproductive effects in birds?
	- Invertivore (aquatic and terrestrial)	
	- Omnivorous wading bird	
	- Piscivorous diving bird	
Mammals	Stable or increasing populations of omnivorous mammals	Is the total daily ingested dose (mg/kg bw-day) of $COPC_E$s greater than doses known to cause effects on the survival, growth and reproduction of mammals?

Table 3-11
Summary of Lines of Evidence for Each Receptor and Assessment Endpoint

Receptor	Assessment Endpoint	Lines of Evidence	Measure of Exposure	Measure of Effect	Comments/Rationale
Benthic Macroinvertebrates	Abundance and diversity of benthic macroinvertebrate communities	Comparison of $COPC_E$ concentrations in sediment to literature-based effects levels	$COPC_E$ Concentrations in sediment (mg/kg dw)	Toxicity reference values for sediment (mg/kg dw)	
		Comparisons of $COPC_E$ concentrations in sediment porewater to literature-based effects levels	$COPC_E$ concentrations in porewater (µg/L)	Toxicity reference values for estuarine and marine waters (µg/L)	Porewater concentrations are modeled using sediment concentrations and Kd or Koc values from the literature (Table 4-5)
Bivalve Molluscs	Stable or increasing populations of bivalves within the site	Comparisons of $COPC_E$ concentrations in clam tissue to literature-based reproductive effect values for molluscs	$COPC_E$ concentrations in clam tissue	Toxicity reference values for invertebrate tissue (ng/kg ww)	
Fish	Stable or increasing populations of fish in the following guilds: benthic omnivore, benthic invertivore, benthic piscivore	Comparison of $COPC_E$ concentrations in surface water to literature-based effects levels	$COPC_E$ concentrations in water (µg/L)	Toxicity reference values for estuarine and marine surface waters ((µg/L)	Surface water concentrations of nickel and BEHP are modeled using sediment concentrations and Kd or Koc values from the literature (Table 4-5)
		Comparison of $COPC_E$ concentrations (metals) in the diet of fish to literature-based effects levels associated with concentrations in the diet of fish	$COPC_E$ concentrations (metals) in food items of fish (mg/kg dw)	Toxicity reference values for concentrations of $COPC_E$s (metals) in food items of fish (mg/kg dw)	
		Comparisons of $COPC_E$ concentrations (PCBs, dioxins, and furans) in fish tissue to literature-based effects levels	$COPC_E$ concentrations (PCBs, dioxins, and furans) in fish tissue (µg/kg lw or ww)	Toxicity reference values for concentrations of $COPC_E$s (PCBs, dioxins, and furans) in fish tissue (ug/kg lw or ww)	
Reptiles	Stable or increasing populations of omnivorous reptiles	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	
Birds	Stable or increasing populations of birds that may be exposed to $COPC_E$s from the site in the following feeding guilds: invertivore (aquatic and terrestrial), omnivorous wading bird, piscivorous diving bird	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	
		Comparison of estimated concentrations of $COPC_E$s (dioxins and furans) in bird eggs to literature-based effects levels for associated with reproductive effects in birds	$COPC_E$ (dioxins and furans) concentration in bird eggs (ng/g ww)	Toxicity reference values for $COPC_E$s (dioxins and furans) in bird eggs (ng/g ww)	Exposure concentrations are estimated using data for concentrations of $COPC_E$s in ingested media (prey and sediment)
Mammals	Stable or increasing populations of omnivorous mammals	Comparison of estimated ingested $COPC_E$ dose to literature-based effects levels expressed on a dose basis	$COPC_E$ doses that account for all ingested media (mg/kg bw-day)	Toxicity reference values for concentrations of COPCEs as ingested doses (mg/kg bw-day)	

Notes

bw = body weight
$COPC_E$ = chemical of potential ecological concern
dw = dry weight

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Birds					
Great Blue Heron	bw	kg	2.2	USEPA (1993)	Average of adult males and females, eastern US, Quinney (1982) in USEPA (2003)
	FIR	kg diet/kg bw-day	0.044	USEPA (1993)	Based on Kushlan (1978); converted from wet weight to dry weight using dietary composition provided by Alexander (1977) and average moisture contents of major prey types provided in USEPA (1993) (FIR dw = FIR ww*(1-% moisture))
	WIR[a]	L/kg bw-day	0.045	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.033	Beyer et al. (1994)	Mallard sediment fraction in diet used as surrogate
	HR	km	2.7	Custer and Galli (2002)	Median flight distance for a Minnesota population
	F_{it}	kg food/kg diet	0		
	F_{ic}	kg food/kg diet	0.01	Alexander (1977)	Fish in diet split equally between large and small fish. Percent vertebrate prey items in diet (5%) reassigned to fish category)
	F_{im}	kg food/kg diet	0	Alexander (1977)	
	F_{fs}	kg food/kg diet	0.495	Alexander (1977)	
	F_{fl}	kg food/kg diet	0.495	Alexander (1977)	
	F_p	kg food/kg diet	0	Alexander (1977)	
Killdeer	bw	kg	0.088	UMMZ (2011)	
	FIR	kg diet/kg bw-day	0.19	Nagy (2001)	Allometric equation for Charadriiformes, dry-weight basis: DMI g/day = 0.522* (g bw) $^{0.769}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	F_s	kg sediment/kg diet	0.10	Beyer et al. (1994)	Value for American woodcock used, as most ecologically similar species available (terrestrial invertivore) in Beyer et al. (1994)
	HR	km^2	0.06	Jackson and Jackson (2000)	Average of home ranges for N=10 in ne CA population; defended breeding territories are considerably smaller and feeding may also take place at much greater distances.
	F_{it}	kg food/kg diet	0.98	Jackson and Jackson (2000)	MO population 98% insects, predominantly terrestrial; Puerto Rico N=20 stomachs 98% animal material, primarily terrestrial invertebrates
	F_{ic}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{im}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{fs}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_{fl}	kg food/kg diet	0	Jackson and Jackson (2000)	
	F_p	kg food/kg diet	0.02	Jackson and Jackson (2000)	2% plant material in gut contents of Puerto Rico study; 1.3% in Missouri study
Neotropic Cormorant	bw	kg	1.3	Telfair and Morrison (2005)	Average of adult males and females
	FIR	kg diet/kg bw-day	0.067	Nagy (2001)	Allometric equation for food intake rates for all birds: DMI g/day = 0.638*g bw $^{0.685}$, divided by bw (kg), multiplied by kg/1000 g to convert to kg diet basis
	WIR[a]	L/kg bw-day	0.054	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.02	Beyer et al. (1994)	Value of <0.02 given for ring-necked duck, as a diving duck is the ecologically most similar species available in Beyer et al. 1994
	HR	N/A	ND	Telfair and Morrison (2005)	No home range information available. Dispersal of juveniles from natal area may be relatively limited, or up to hundreds of kilometers.
	F_{it}	kg food/kg diet	0		Diet almost entirely comprised of fish in local study. Primarily fish <8 mm taken (see exp areas worksheet). Small proportion of shrimp in diet added into fish category, as this is primarily a pelagic invertebrate pathway that would not be well-represented by benthic invertebrate tissue data
	F_{ic}	kg food/kg diet	0	King (1989)	
	F_{im}	kg food/kg diet	0	King (1989)	
	F_{fs}	kg food/kg diet	1	King (1989)	
	F_{fl}	kg food/kg diet	0	King (1989)	
	F_p	kg food/kg diet	0	King (1989)	

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Spotted Sandpiper	bw	kg	0.043	USEPA (1993)	Average of males and females, Maxson and Oring (1980)
	FIR	kg diet/kg bw-day	0.22	Nagy (2001)	Allometric equation for Charadriiformes, dry-weight basis: DMI g/day = 0.522* (g bw) $^{0.705}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	WIR[a]	L/kg bw-day	0.17	Calder and Braun (1983) in USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.18	Beyer et al. (1994)	Average of data for four sandpiper species (range 7 to 30%)
	HR	km	1.5	Macwhirter et al. (2002)	No HR information for spotted sandpiper; this value is a home range for sanderling, a similarly sized invertivorous shorebird that winters in coastal Texas.
	F_{it}	kg food/kg diet	0	USEPA (1993)	
	F_{ic}	kg food/kg diet	0.5		
	F_{im}	kg food/kg diet	0.5		
	F_{fs}	kg food/kg diet	0		
	F_{fl}	kg food/kg diet	0		
	F_p	kg food/kg diet	0		
White-Faced Ibis[b]	HR	km^2	12	GBBO (2012)	Minimum recommended habitat patch size based on expert opinion and limited home range information
Bald Eagle[b]	HR	km^2	14.5; 145	Buehler (2012)	Average home ranges for breeding and non-breeding (wintering) populations, respectively, of bald eagles
Brown Pelican[b]	HR	N/A	see notes		No home range information available. Foraging radius from nesting sites described as within 20 km radius of nesting colony during breeding season, up to 75 km from nearest land during non-breeding season
Mammals					
Marsh Rice Rat	bw	kg	0.051	Davis and Schmidly (1994)	Average of range of adult weights
	FIR	kg diet/kg bw-day	0.19	Nagy (2001)	Allometric equation for mesic rodents, dry-weight basis: DMI g/day = 0.614* (g bw) $^{0.705}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	F_s	kg sediment/kg diet	0.02	Beyer et al. (1994)	Value of < 0.02 given for white-footed mouse, most ecologically similar mammal available in Beyer et al. 1994
	HR	km	0.075	Wolfe (1982)	Average of Maryland (75 m) and Florida (68 and 82m) range lengths
	F_{it}	kg food/kg diet	0		Estimated assignments based on Wolfe's summary of multiple studies, which indicates multiple food sources, with roughly equal amounts of plant and animal materials. Small fish, clams, crabs, snails, bird eggs among common gut contents.
	F_{ic}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{im}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{fs}	kg food/kg diet	0.2	Wolfe (1982)	
	F_{fl}	kg food/kg diet	0		
	F_p	kg food/kg diet	0.4	Wolfe (1982)	
Raccoon	bw	kg	5.1	USEPA (1993)	Average of adult males and females from an Alabama population and a Missouri population
	FIR	kg diet/kg bw-day	0.041	Nagy (2001)	Allometric equation for placental mammals,: DMI g dw/day= 0.299 (g bw) 0.767, divided by bw in kg and kg/1,000 g to convert to kg diet basis
	F_s	kg sediment/kg diet	0.094	Beyer et al. (1994)	
	HR	km^2	0.52	USEPA (1993)	Average of male and female year-round ranges on a Georgia coastal island (Lotze 1979)
	F_{it}	kg food/kg diet	0.05	Alexander (1977)	Dietary composition by % of wet weight for 29 raccoons: % vertebrates in diet were reassigned to invertebrates and fish categories; percent unidentified material reassigned to plant category
	F_{ic}	kg food/kg diet	0.24	Alexander (1977)	
	F_{im}	kg food/kg diet	0.05	Alexander (1977)	
	F_{fs}	kg food/kg diet	0.20	Alexander (1977)	
	F_{fl}	kg food/kg diet	0.20	Alexander (1977)	
	F_p	kg food/kg diet	0.26	Alexander (1977)	

Table 3-12
Receptor-Specific Life History Parameters for the WildlifeExposure Model

Species	Parameter	Units	Value	Source	Notes
Reptiles					
Alligator Snapping Turtle	bw	kg	51.5	National Geographic (2011)	Average of bw for male and female alligator snapping turtles of 80 and 23 kg, respectively.
	FIR	kg diet/kg bw-day	0.01	Nagy (2001)	Allometric equation for carnivorous reptiles, dry-weight basis: DMI g/day = 0.00865* (g bw) $^{0.963}$, divided by kg bw and converted to kg diet basis by * kg/1,000 g
	WIR[a]	L/kg bw-day	0.02	USEPA (1993)	Water ingestion rate
	F_s	kg sediment/kg diet	0.05	Beyer et al. (1994)	Value for sediment in the diet of box turtle, as snapping turtle data not available.
	HR	km	0.778	Riedle (2008)	
	F_{it}	kg food/kg diet	0		Assignments to prey categories based on indices of relative importance for invertebrates calculated from Elsey. Vertebrates in diet were reassigned to fish (fish as proportion of diet split equally between large and small); unidentified material was reassigned to plant category.
	F_{ic}	kg food/kg diet	0.03	Elsey (2006)	
	F_{im}	kg food/kg diet	0.01	Elsey (2006)	
	F_{fs}	kg food/kg diet	0.35	Elsey (2006)	
	F_{fl}	kg food/kg diet	0.35	Elsey (2006)	
	F_p	kg food/kg diet	0.26	Elsey (2006)	

Notes

AUF= area use factor
bw = body weight (kg)
Fs = fraction of the diet that is sediment
F_{fs} = fraction of the diet consisting of small fish (kg fish/kg food)
F_{fl} = fraction of the diet consisting of large fish (kg fish/kg food)
F_{it} = fraction of the diet consisting of terrestrial invertebrates (kg invertebrates/kg food)
F_{ic}= fraction of diet consisting of crustacea (kg invertebrates/kg food)
F_{im} = fraction of the diet consisting of molluscs (kg invertebrates/kg food)
F_p = fraction of the diet consisting of plants (kg plants/kg food)
FIR = food ingestion rate (kg food dw/day)
HR = home range
WIR = water ingestion rate

a - allometric equation for birds, WIR (L/day) = (0.059*BW(kg)0.67)/kg bw
b - state or federally listed species, evaluated in cases where risk to surrogate receptors $HQ_N \geq 1$

Table 4-1

Partition Coefficients for Chemicals of Potential Ecological Concern

Chemical	K_d [a] (L/kg)	K_{OC} [b] (L/kg)
Metals		
Barium	41	NA
Cadmium	75	NA
Cobalt	45	NA
Copper	35	NA
Lead	900	NA
Magnesium	5	NA
Manganese	65	NA
Mercury	52	NA
Nickel	65	NA
Thallium	71	NA
Zinc	62	NA
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	NA	120,000
Carbazole	NA	9,160
Phenol	NA	187

Notes

NA = not applicable

a - Soil-water partition coefficient from the Risk Assessment Information System (USDOE 2012)

b - Organic carbon partition coefficient from the Risk Assessment Information System (USDOE 2012)

Table 4-2

Summary Statistics for Estimated Porewater Concentrations of $COPC_E$s

$COPC_E$	Units	N	Min	Mean	Max
Metals					
Cobalt	mg/L	97	0.0044	0.092	0.30
Manganese	mg/L	97	0.025	3.7	23
Thallium	mg/L	97	0.0028	0.019	0.049
Semivolatile Organic Compounds					
Bis(2-ethylhexyl)phthalate	µg/L	97	0.00804	0.0809	1.85

Notes

$COPC_E$ = chemical of potential ecological concern

Table 4-3
Receptor-Specific Dietary Assumptions for Fish

Species	Parameter	Literature-Based Value	Modeled Value[a]	Sources for Dietary Estimates	Notes
Gulf killifish	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for Pacific staghorn sculpin adopted for Lower Duwamish Waterway Group BERA (Windward 2007)
	F_{it}	0.19		Hassan-Williams and Bonner (2012); USGS (2009)	Omnivorous: feeds throughout the water column, consuming benthic algae, vascular plants, grass shrimp, microcrustaceans, terrestrial insects that fall onto the water surface, mosquito larvae and pupae, bivalve molluscs, and small fishes (e.g., killifishes and anchovies). For modeling exposure, terrestrial invertebrate portion of diet reassigned to aquatic invertebrates, and plants component reassigned to invertebrate and fish components of diet.
	F_{ap}	0.2			
	F_{ic}	0.2	0.36		
	F_{im}	0.2	0.36		
	F_{fs}	0.2	0.27		
Black drum	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for English sole and Pacific staghorn sculpin adopted for Lower Duwamish Waterway Group BERA (Windward 2007).
	F_{ap}	0.04		TPWD (2012a); Sutter et al. (1982); LSU (2012); Smithsonian (2012)	Benthic invertivore: young black drum (< 20 cm) feed on marine worms and small fish, shrimp and crab and larger young (8 to 20 cm) eat small fish (36 percent) and polychaetes (32 percent). Larger drum (> 20 cm) consume molluscs, small crabs, worms and algae. In Texas estuaries, the dominant food of black drum (21 to 50 cm) is the mollusc *Mulinia* sp. (33 percent). The largest drum ate mostly molluscs (74 percent) and crabs (16 percent). For modeling exposure, polychaetes and plants portion of diet reassigned to aquatic invertebrates.
	F_{ip}	0.05			
	F_{ic}	0.16	0.176		
	F_{im}	0.7	0.814		
	F_{fs}	0			
Southern flounder	Fs	0.01	0.01	Windward (2007)	Sediment ingestion portion of diet for English sole adopted for Lower Duwamish Waterway Group BERA (Windward 2007).
	F_{it}	0		Hassan-Williams and Bonner (2012), TPWD (2012b)	Benthic piscivore: small fishes (e.g., anchovies, juvenile striped mullet, menhadens, Atlantic croaker, spot, pinfish, and fat sleeper) and to a lesser extent, crustaceans (e.g., mysids, isopods, amphipods, penaeid shrimp, and portunid crabs) constitute most of the southern flounder diet. Ninety-five percent of the food items of juvenile southern flounder (10 to 150 mm) from Texas consisted of invertebrates. Juvenile southern flounder (>80 mm) consume progressively larger food items as they grow. Fish make up 70 percent of the adult (>150 mm) diet.
	F_{ic}	0.29	0.29		
	F_{im}	0			
	F_{fs}	0.7	0.7		

Notes

Fs = fraction (unitless) of the diet that is sediment
F_{it} = fraction (unitless) of the diet consisting of terrestrial invertebrates
F_{ap} - fraction (unitless) of diet consisting of aquatic plants
F_{ip} = fraction (unitless) of diet consisting of polychaetes
F_{ic}= fraction (unitless) of diet consisting of crustacea
F_{im} = fraction (unitless) of the diet consisting of molluscs
F_{fs} = fraction (unitless) of the diet consisting of small fish

a - The modeled value reassigns the literature-based proportion of prey that is in a category for which empirical tissue data are not available, to a category of prey which is ecologically similar and for which for which empirical data are available. A category for which empirical tissue data are not available is reassigned to categories for which data are available in the modeled diet, weighted by their relative abundance (e.g., polychaetes in the black drum diet are reassigned to crustacea as % polychaetes * (% crustacea in black drum diet/(%crustacea +%molluscs)); and to molluscs as %polychaetes*(%molluscs in black drum diet/(%molluscs+%crustacea))

Table 4-4
Surface-Area Weighted Average Concentrations of $COPC_E$s in Sediments of the Site and Estimated Concentrations in Surface Water

Analyte	Sediment SWAC	K_d[a] (L/kg)	K_{OC}[a] (L/kg)	Estimated Concentration in Surface Water[b]
SVOCs	**µg/kg OC**			**µg/L**
Bis(2-ethylhexyl)phthalate	16,400	N/A	120,000	0.14
Metals	**mg/kg dw**			**mg/L**
Cadmium	0.440	75	N/A	0.00586
Copper	11.4	35	N/A	0.326
Lead	11.8	900	N/A	0.0131
Mercury	0.0495	52	N/A	0.0010
Nickel	6.26	65	N/A	0.096
Zinc	55.0	62	N/A	0.887

Notes

$COPC_E$ = chemical of potential ecological concern
SVOC = semivolatile organic compound
SWAC = surface area-weighted average concentration

a - See Table 4-1 for source of Kd and Koc values.
b - These gross and highly conservative estimates of surface water chemical concentrations are calculated as sediment SWAC ÷ K_d for inorganics and SWAC ÷ K_{oc} for organics, per Eqn. 4-2.

Table 4-5
Total PCBs Concentrations in Whole Fish from Site and Background

Hardhead Catfish		*Gulf Killifish*	
	Total PCBs		**Total PCBs**
Sample ID	**(µg/kg ww)**	**Sample ID**	**(µg/kg ww)**
FCA1			
SJFCA1-LF1	588	GK-TTR2-1	33.5
SJFCA1-LF6	664	GK-TTR2-2	40.1
SJFCA1-LF10	759		
FCA2			
SJFCA2-LF1	793	GK-TTR3-1	187
SJFCA2-LF4	647	GK-TTR3-2	191
SJFCA2-LF8	563	GK-TTR4-1	24.2
SJFCA2-LF10	286	GK-TTR4-2	19.4
		GK-TTR5-1	44
		GK-TTR5-2	32.7
FCA3			
SJFCA3-LF1	469	GK-TTR6-1	51.9
SJFCA3-LF6	750	GK-TTR6-2	28.9
SJFCA3-LF10	942		
Background			
SJFCACB-LF1	137	GK-TTR7-1	13.1
SJFCACB-LF2	347	GK-TTR7-2	12.3
SJFCACB-LF4	163	GK-TTR7-3	13.9
SJFCACB-LF5	206	GK-TTR7-4	15.5
SJFCACB-LF6	251	GK-TTR8-1	12.2
SJFCACB-LF8	192	GK-TTR8-2	13
SJFCACB-LF9	460	GK-TTR8-3	11.9
SJFCACB-LF10	412	GK-TTR8-4	14.1

Table 4-6

Dioxins, Furans, and PCBs in Whole Fish Expressed as $TEQ_{DFP,F}$

	$TEQ_{DF,F}$ [a] ng/kg lw		$TEQ_{DFP,F}$ [b] ng/kg lw				$TEQ_{DF,F}$ [a] ng/kg lw		$TEQ_{DFP,F}$ [b] ng/kg lw	
Hardhead Catfish						***Gulf Killifish***				
Sample ID						**Sample ID**				
FCA1										
SJFCA1-LF1	**323**	J	**330**	J		GK-TTR2-1	147	U	197	J
SJFCA1-LF6	264	J	271	J		GK-TTR2-2	8.72	U	12	J
SJFCA1-LF10	**367**	J	**381**	J						
FCA2										
SJFCA2-LF1	315	J	**321**	J		GK-TTR3-1	137	J	142	J
SJFCA2-LF4	314	J	**326**	J		GK-TTR3-2	265	J	270	J
SJFCA2-LF8	205	J	212	J		GK-TTR4-1	5.98	J		J
SJFCA2-LF10	183	J	188	J		GK-TTR4-2	307	U	**503**	J
						GK-TTR5-1	160	J	169	J
						GK-TTR5-2	11	J	12.9	J
FCA3										
SJFCA3-LF1	221	J	229	J		GK-TTR6-1	3.38	J	3.88	J
SJFCA3-LF6	286	J	291	J		GK-TTR6-2	4.87	J	5.28	J
SJFCA3-LF10	**373**	J	**381**	J						
Background										
SJFCACB-LF1	25.7	J	26.6	J		GK-TTR7-1	4.29	J	4.75	J
SJFCACB-LF2	36.6	J	39.4	J		GK-TTR7-2	3.71	J	4.46	J
SJFCACB-LF4	23.2	J	26.6	J		GK-TTR7-3	3.52	J	3.74	J
SJFCACB-LF5	44.3	J	48.1	J		GK-TTR7-4	15.7	J	17.5	J
SJFCACB-LF6	41.4	J	45.5	J		GK-TTR8-1	2.11	U	2.95	J
SJFCACB-LF8	36.9	J	40.3	J		GK-TTR8-2	0.857	J	1.12	J
SJFCACB-LF9	68.0	J	76.8	J		GK-TTR8-3	3.67	J	4.11	J
SJFCACB-LF10	47.5	J	54.1	J		GK-TTR8-4	3.04	J	3.84	J

Notes

Bold indicates that the concentration is greater than that considered protective of 95 percent of fish species

lw = lipid weight

J = One or more congener used in calculation of TEQ was not detected

FCA = fish collection area

a - Toxicity equivalent for dioxins and furans calculated using fish toxicity equivalency factors with nondetects set at one-half the detection limit.

b - Toxicity equivalent for dioxins, furans, and polychlorinated biphenyls calculated using fish toxicity equivalency factorswith nondetects set at one-half the detection limit.

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Gulf Killifish - Transect 1 and 2 (FCA 1)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.005	0.099	0.108	0.093	0.103	0.002	0.003	0.199	0.219
Copper	0.1	0.1	16.3	20.1	6.3	6.7	1.4	1.5	24.2	28.4
Mercury	0.0003	0.0004	0.03	0.04	0.03	0.04	0.03	0.04	0.10	0.11
Zinc	0.4	0.5	42.3	44.1	38.9	42.0	44.8	46.9	126	134

$COPC_E$	Gulf Killifish - Transect 3 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.003	0.010	0.115	0.133	0.099	0.110	0.002	0.003	0.220	0.256
Copper	0.1	0.3	17.4	21.2	13.8	15.0	1.5	1.6	32.8	38.1
Mercury	0.003	0.01	0.02	0.03	0.05	0.05	0.07	0.09	0.14	0.17
Zinc	0.4	1.1	40.7	44.4	35.9	37.6	45.3	46.6	122	130

$COPC_E$	Gulf Killifish - Transect 4 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.002	0.006	0.115	0.133	0.082	0.088	0.002	0.002	0.201	0.229
Copper	0.0173	0.089	17.4		6.4	7.2	1.4	1.7	25.2	30.2
Mercury	0.0002	0.0005	0.02	0.03	0.03	0.04	0.06	0.10	0.11	0.16
Zinc	0.1	0.5	40.7	44.4	30.7	33.6	45.2	45.6	117	124

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Gulf Killifish - Transect 5 (FCA 2)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.001	0.004	0.115	0.133	0.076	0.082	0.002	0.003	0.194	0.222
Copper	0.1	0.1	17.4	21.2	6.4	7.7	1.6	1.9	25.5	30.8
Mercury	0.0001	0.0003	0.02	0.03	0.02	0.02	0.04	0.04	0.08	0.09
Zinc	0.2	0.5	40.7	44.4	34.8	37.0	45.6	46.7	121	129

$COPC_E$	Gulf Killifish - Transect 6 (FCA 3)									
	Sediment		Prey - Crustacea		Prey - Molluscs		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.001	0.001	0.079	0.099	0.095	0.104	0.002	0.002	0.177	0.206
Copper	0.01	0.04	16.4	17.7	11.6	11.8	1.6	1.7	29.6	31.2
Mercury	0.0001	0.0001	0.03	0.04	0.05	0.06	0.07	0.08	0.15	0.18
Zinc	0.03	0.1	38.2	40.5	33.7	35.8	50.8	52.0	123	128

$COPC_E$	Black Drum (Site-wide)							
	Sediment		Prey - Crustacea		Prey - Molluscs		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.006	0.048	0.054	0.200	0.209	0.252	0.268
Copper	0.1	0.2	8.1	9.0	20.0	26.5	28.2	35.8
Mercury	0.001	0.003	0.013	0.015	0.078	0.088	0.092	0.106
Zinc	0.3	1.0	19.7	20.6	78.0	80.8	98.1	102

Table 4-7

Weighted Concentrations[a] of $COPC_E$s (mg/kg dw) in the Diets of Fish

$COPC_E$	Southern Flounder (Site-wide)							
	Sediment		Prey - Crustacea		Prey - Small fish		Total Diet[b]	
	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	0.004	0.006	0.078	0.088	0.006	0.006	0.088	0.101
Copper	0.1	0.2	13.4	14.9	4.0	4.4	17.5	19.5
Mercury	0.001	0.003	0.022	0.025	0.142	0.184	0.164	0.211
Zinc	0.3	1.0	32.4	33.9	122	126	155	161

Notes

CT = central tendency

RM = reasonable maximum

a - Weighted concentrations are the product of the transect-specific EPC for the prey item (Table C-2) and the prey item's estimated fraction of the total diet as described in Table 4-3. For example, the CT of the weighted concentration of cadmium in blue crab in the diet of gulf killifish in FCA1 is 0.273 mg/kg (CT EPC of cadmium in crab) x 0.36 (the fraction of crustacea in the diet of killifish) = 0.099 mg/kg dw.

b - Total diet is the sum of the weighted concentrations of prey and sediment.

Table 4-8
Assumptions for Parameterizing the Wildlife Exposure Model

			Area of the Site/Data Source Used for Model Parameterization									
Taxon	Feeding Guild	Receptor	Aquatic/ Terrestrial	Exposure Unit	Surface Water	Sediments (0- to 6-inch depth)	Soils (0- to 6-inch depth)	Fish	Terrestrial Invertebrates	Benthic Invertebrates	Plants	Notes
Birds	Piscivore (wading)	Great blue heron[a]	Aquatic	All accessible shorelines of the site	Estimated from sediment SWACs [b]	All site shoreline sediments	N/A	Small and large fish	N/A	Shoreline invertebrate tissue samples	N/A	Great blue heron is limited to fish < 25 cm, but will use large fish (30 to 45 cm) to estimate exposure to other receptors that may ingest this size range.
	Invertivore (terrestrial)	Killdeer	Terrestrial	All upland areas of the site north of I-10	N/A	N/A	Soil data for site north of I-10	N/A	BAFs from upland soils N of I-10 for non-dioxin COPCs; regression approach for dioxins and furans (Appendix D)	N/A	N/A	
	Piscivore (diving)	Neotropic cormorant	Aquatic	All aquatic areas of the site	Estimated from sediment SWACs [b]	Site-wide sediments	N/A	Small fish: <8 cm TL	N/A	N/A	N/A	Fold pelagic invertebrates (2% of diet) into fish so 100% fish modeled in diet
	Invertivore (probing)	Spotted sandpiper	Aquatic	All accessible shorelines of the site	Estimated from sediment SWACs [b]	All site shoreline sediments	N/A	N/A	N/A	Shoreline invertebrate tissue samples	N/A	
Mammals	Omnivore	Marsh rice rat	Aquatic	All accessible shorelines of the site	N/A	All site shoreline sediments	N/A	Small fish: <8 cm TL	N/A	Shoreline invertebrate tissue samples	BAFs from shoreline sediments	
	Omnivore	Raccoon	Aquatic and terrestrial	Non-island uplands and shorelines of accessible areas of the peninsula	N/A	Shoreline sediments of the peninsula	Soils of the peninsula	Small fish from the peninsula shoreline	BAFs from upland peninsula soils for non-dioxin COPCs; regression approach for dioxins (Appendix D)	Peninsula shoreline invertebrate tissue samples	BAFs from peninsula soils	Assumes receptor uses both upland and shorelines for foraging: soil and sediment ingestion each receive one-half of incidental ingestion rate
Reptiles	Omnivore	Alligator snapping turtle	Aquatic	All aquatic areas of the site	Estimated from sediment SWACs [b]	Site-wide sediments	N/A	Site-wide: all fish	N/A	Site-wide: all aquatic invertebrates	BAFs from shoreline sediments	

Notes

BAF = bioaccumulation factor
COPC = chemical of potential concern
N/A = not applicable (no exposure to this medium is expected)
SWAC = surface area-weighted average concentration
a - Surrogate receptor for bald eagle, for which assumptions are identical to this receptor except home range and area use factor (see Table 4-12)
b - Except for dioxins and furans, for which empirical data are used
c - Surrogate receptor for white-faced ibis, for which assumptions are identical to this receptor except home range and area use factor (see Table 4-12)

Table 4-9
Relative Bioavailability Adjustment Factors for TCDD in Soil, Sediment, and Food Ingested by Birds

Medium	Relative Bioavailability Adjustment Factor
Invertebrates[a]	0.44
Soils	0.33
Sediments[b]	0.41

Source
Nosek et al. (1992a)

Notes
a - Average of percent absorption from homogenate of earthworms and homogenate of crickets
b - Percent absorption from paper mill sludge solids

Table 4-10
Bioaccumulation Relationships for Soil-to-Invertebrates and Soil-to-Plant Tissue

Chemical	Concentration in Invertebrate Tissue [a,b] (mg/kg dw)	Concentration in Plant Tissue [a,b] (mg/kg dw)
Dioxins and Furans		
Dioxins and Furans	See Appendix D	0 [c]
Polychlorinated Biphenyls		
Polychlorinated Biphenyls	$Cs^{1.361} * e^{1.41}$ [d]	0 [c]
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	0 [e]	0 [c]
Metals		
Cadmium	$Cs^{0.795} * e^{2.114}$	$Cs^{0.546}/e^{0.475}$
Chromium	Cs*0.306	Cs*0.041
Cobalt	Cs * 0.122	Cs * 0.0075
Copper	Cs * 0.515	$Cs^{0.394} * e^{0.668}$
Lead	$Cs^{0.807}/e^{0.218}$	$Cs^{0.561}/e^{1.328}$
Mercury	Cs*3.1 for Cs ≤1.5 mg/kg; [f] Cs*0.7 for Cs > 1.5 mg/kg	Cs*0.0375 [g]
Nickel	(Cs*0.02)/0.16 [h]	$Cs^{0.748}/e^{2.223}$
Vanadium	Cs * 0.042	Cs * 0.00485
Zinc	$Cs^{0.328} * e^{4.449}$	$Cs^{0.554} * e^{1.575}$

Notes

BAF = bioaccumulation factor

Cs = concentration in soil (mg/kg)

dw = dry weight

a - Unless otherwise indicated, the source for values in this column is USEPA (2007c) Attachment 4-1: Exposure Factors and Bioaccumulation Models for Derivation of Wildlife EcoSSLs (Table 4a).

b - Natural log equations were transformed as follows:

$\ln(y) = a*\ln(x)-b$, transformed to $y = x^a/e^b$; or

$\ln(y) = a*\ln(x)+b$, transformed to $y = x^a*e^b$

c - Dioxins, PCBs, and BEHP are low-solubility, high molecular weight compounds which have a negligible potential for uptake into plant tissues (Staples et al. 1997; Bacci et al. 1992; McCrady et al. 1990, 1993); therefore, a BAF of zero is used for these $COPC_E$s.

d - Sample et al. (1998). Regression equation from Table 12 for total PCBs.

e - BEHP does not bioaccumulate in invertebrate tissue at environmentally realistic concentrations in soil (Staples et al. 1997).

f - Based on differential uptake by earthworms depending on soil concentrations: a higher BAF for soils with lower mercury concentrations, and a lower BAF for soils with higher mercury concentrations (Burton et al. 2006)

g - Recommended soil to plant bioconcentration factor from Table C-2 for mercuric chloride in USEPA (1999b).

h - Recommended soil-to-invertebrate bioconcentration factor from Table C-1 in USEPA (1999b). Because the BCF provided by USEPA (1999b) is on basis of kg dw soil/kg ww tissue, the resulting value is converted to dw tissue basis by dividing by (1-moisture content), where moisture content = 0.86 (USEPA 1993).

Table 4-11

Regression Equations Used to Estimate Dioxin and Furan Congener Concentrations in Terrestrial Invertebrate Tissue

Congener	Equation
2,3,7,8-TCDD	$\exp(-2.49 + 0.819*(\ln(C_{s2,3,7,8\text{-TCDD}}))$
1,2,3,7,8-PCDD	$\exp(-5.92+0.516*(\ln(C_{s1,2,3,7,8\text{-PCDD}}))$
1,2,3,4,7,8-HxCDD	$0.430*C_{e1,2,3,7,8,9\text{-HxCDD}}$
1,2,3,6,7,8-HxCDD	$\exp(-3.42+0.664*(\ln(C_{s1,2,3,6,7,8\text{-HxCDD}}))$
1,2,3,7,8,9-HxCDD	$\exp(-5.04+0.55*(\ln(C_{s1,2,3,7,8,9\text{-HxCDD}}))$
1,2,3,4,6,7,8-HpCDD	$\exp(-3.91+0.479*(\ln(C_{s1,2,3,4,6,7,8\text{-HpCDD}}))$
OCDD	$8.02*C_{e1,2,3,4,6,7,8\text{-HpCDD}}$
2,3,7,8-TCDF	$0.120*C_{e1,2,3,6,7,8\text{-HxCDD}}$ [a]; $0.250*C_{e1,2,3,4,7,8\text{-HxCDF}}$ [b]
1,2,3,7,8-PCDF	$\exp(-4.86+0.593*(\ln(C_{s1,2,3,7,8\text{-PCDF}}))$
2,3,4,7,8-PCDF	$0.108*C_{e1,2,3,6,7,8\text{-HxCDD}}$
1,2,3,4,7,8-HxCDF	$\exp(-4.29+0.616*(\ln(C_{s1,2,3,4,7,8\text{-HxCDF}}))$
1,2,3,6,7,8-HxCDF	$\exp(-4.50+0.609*(\ln(C_{s1,2,3,6,7,8\text{-HxCDF}}))$
1,2,3,7,8,9-HxCDF	$\exp(-5.74+0.671*(\ln(C_{s1,2,3,7,8,9\text{-HxCDF}}))$
2,3,4,6,7,8-HxCDF	$\exp(-5.22+0.576*(\ln(C_{s2,3,4,6,7,8\text{-HxCDF}}))$
1,2,3,4,6,7,8-HpCDF	$\exp(-3.69+0.593*(\ln(C_{s1,2,3,4,6,7,8\text{-HpCDF}}))$
1,2,3,4,7,8,9-HpCDF	$0.723*C_{e1,2,3,4,7,8\text{-HxCDF}}$
OCDF	$0.603*C_{e1,2,3,4,6,7,8\text{-HpCDD}}$

Notes

$C_{scongener}$= concentration of the given congener in soil

$C_{econgener}$ = concentration of the given congener in earthworms

a - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples outside of the impoundments.

b - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples inside of the impoundments.

Table 4-12
Area Use Factors Used to Evaluate Exposure of Wildlife Receptors

	Alligator Snapping Turtle	Neotropic Cormorant	Great Blue Heron	Spotted Sandpiper	Marsh Rice Rat	Raccoon	Killdeer	White-Faced Ibis [a]	Brown Pelican[a]	Bald Eagle [a]	
Exposure Unit [b]	All aquatic shorelines of the site	All aquatic areas of the site	All aquatic shorelines of the site	All aquatic shorelines of the site	All aquatic shorelines of the site	Terrestrial area of the peninsula	Terrestrial area north of I-10	All aquatic shorelines of the site [c]	All aquatic areas of the site	All aquatic shorelines of the site	
										breeding [d]	wintering [d]
Estimated Size of Exposure Unit	37.61 km	2.52 km^2	37.6 km	37.6 km	37.6 km	0.36 km^2	0.13 km^2	0.38 km^2	2.52 km^2	2.52 km^2	2.52 km^2
Home Range [e]	0.778 km	ND	2.7 km	1.5 km	0.075 km	0.52 km^2	0.06 km^2	12 km^2	1,257 km^2	14.5 km^2	125 km^2
AUF [f]	1	1	1	1	1	0.68	1	0.03	0.002	0.17	0.02

Notes

AUF = area use factor

a - Listed species; all other life history parameters are based on surrogate receptors. which are spotted sandpiper for ibis and great blue heron for bald eagle.

b - The exposure unit is calculated in units that match the units of the home range so that an AUF may be calculated. See Figures 4-13 through 4-17 for illustrations of these exposure units.

c - Home range for white-faced ibis is given on a km^2 basis, which was converted to relevant habitat area at the site by multiplying total shoreline length by a width of 10 m around the shoreline based or shallow water foraging strategy of this species (Safran et al. 2000).

d - Bald eagles have primarily been noted as wintering in site vicinity, but their breeding distribution may include the site vicinity, so AUFs are calculated for both breeding and non-breeding eagles.

e - Receptor home ranges are further described in Table 3-12.

f - Receptors whose home range is less than the exposure unit are assigned an AUF of 1; for receptors lacking home range data, an AUF of 1 is assumed.

Table 4-13
Daily Ingestion Rates of $COPC_E$s for Aquatic and Upland (North of I-10) Wildlife Receptors

	Ingestion Rate (mg/kg bw-day)																					
	Great Blue Heron		Spotted Sandpiper		Neotropic Cormorant		Killdeer		Marsh Rice Rat		Raccoon		Alligator Snapping Turtle		White-Faced Ibis		Bald Eagle: Breeding		Bald Eagle: Wintering		Brown Pelican	
Analyte	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cadmium	0.0015	0.0021	0.071	0.087	0.0013	0.0016	0.68	1.2	0.048	0.061	0.0059	0.0082	8.3×10^{-4}	0.0011	0.0022	0.0027	2.7×10^{-4}	3.6×10^{-4}	2.7×10^{-5}	3.6×10^{-5}	2.7×10^{-6}	3.2×10^{-6}
Copper	0.21	0.27	8.1	10	0.41	0.46	1.0	4.9	3.1	3.7	0.43	0.60	0.033	0.042	0.25	0.32	0.036	0.047	0.0036	0.0047	0.0008	0.0009
Mercury	0.010	0.013	0.027	0.042	0.014	0.018	0.19	0.54	0.016	0.021	0.0048	0.0089	9.7×10^{-4}	0.0012	0.0008	0.0013	0.0018	0.0023	0.00018	0.00023	3.0×10^{-5}	4.0×10^{-5}
Nickel	0.074	0.13	1.7	2.1	0.14	0.19	0.26	0.92	0.63	0.78	0.061	0.12	0.010	0.016	0.053	0.065	0.013	0.022	0.0013	0.0022	0.0003	0.0004
Zinc	18	20	24	28	12	12	56	100	17	21	6.0	8.1	1.8	2.0	0.75	0.89	3.1	3.5	0.31	0.35	0.02	0.02
Bis(2-ethylhexyl)phthalate	0.026	0.033	0.21	0.26	0.029	0.029	8.2×10^{-4}	0.0096	0.089	0.11	0.014	0.019	0.0026	0.0033	0.0066	0.0082	0.0046	0.0058	0.00046	0.00057	6.0×10^{-5}	6.0×10^{-5}
$TEQ_{DF, B}$ [a]	6.8×10^{-6}	1.6×10^{-5}	1.7×10^{-4}	3.8×10^{-4}	1.5×10^{-6}	7.8×10^{-6}	4.3×10^{-5}	1.3×10^{-4}	N/A	N/A	N/A	N/A	4.4×10^{-7}	1.2×10^{-6}	4.6×10^{-6}	1.2×10^{-5}	1.2×10^{-6}	2.7×10^{-6}	1.2×10^{-7}	2.7×10^{-7}	3.0×10^{-9}	1.6×10^{-8}
$TEQ_{P, B}$ [b]	1.0×10^{-6}	1.3×10^{-6}	5.7×10^{-6}	8.4×10^{-6}	6.3×10^{-7}	1.2×10^{-6}	4.2×10^{-8}	5.6×10^{-8}	N/A	N/A	N/A	N/A	9.7×10^{-8}	1.3×10^{-7}	1.8×10^{-7}	2.6×10^{-7}	1.8×10^{-7}	2.3×10^{-7}	1.8×10^{-8}	2.8×10^{-8}	1.3×10^{-9}	2.3×10^{-9}
$TEQ_{DF, M}$ [c]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	6.3×10^{-6}	1.7×10^{-5}	3.5×10^{-6}	8.9×10^{-6}	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
$TEQ_{P, M}$ [d]	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	3.4×10^{-7}	6.8×10^{-7}	1.8×10^{-7}	2.5×10^{-7}	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A	N/A
Total PCBs	0.057	0.098	0.59	1.4	0.015	0.038	0.030	0.038	0.073	0.17	0.029	0.060	0.0037	0.0055	0.018	0.045	0.010	0.017	0.0010	0.0017	3.0×10^{-5}	0.0001

Notes

CT = central tendency

RM = reasonable maximum

a - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

b - Toxicity equivalent for dioxin-like PCBs calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

c - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection

d - Toxicity equivalent for dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.

Table 4-14
Regression Models Developed by Elliott et al. (2001) for Predicting Dioxin and Furan Concentrations in Bird Eggs from Prey Fish of Birds

Congener	r^2	Slope	Intercept	p-value
2,3,7,8-TCDD	0.848	0.869	1.484	0.004
∑ PeCDD	0.904	0.647	1.832	0.002
∑HxCDD	0.917	0.662	1.757	<0.001
2,3,7,8-TCDF	0.628	0.407	0.333	0.07
∑PeCDF	0.847	0.741	1.4	0.008

Source

Elliott et al. (2001); Equation: log (egg [ ]) = slope x log (prey [ ]) + Intercept

Table 4-15

Sample Location Identification and Associated Dioxin and Furan Concentrations for Prey and Sediment Media at the Site and for Post-TCRA and Background Scenarios

	Ingestion Fraction for Each Receptor						Congener (ng/kg ww)							
Dietary Source	**Cormorant**	**Heron**	**Sandpiper**	**Scenario**	**Mode**	**Sample ID**	TCDD	ΣPeCDD	ΣHxCDD	ΣHpCDD	TCDF	ΣPeCDF	ΣHxCDF	ΣHpCFD
Blue crab	NA	0.01	0.5	Site	CT	SJFCA2-CR6	1.60	0.237	0.718	0.612	5.82	0.714	0.0872	0.0200
					RM	SJFCA1-CR6	3.34	1.16	1.67	0.864	10.2	2.61	0.259	0.0216
				Background	CT	SJFCACB-CR6	0.0668	0.0374	0.384	0.635	0.281	0.117	0.0186	0.0216
					RM	SJFCACB-CR1	0.124	0.0322	0.461	0.424	0.251	0.0840	0.0361	0.0153
Common rangia	NA	NA	0.5	Site	CT	CL-TTR5-001	1.58	0.0256	0.0379	0.403	7.73	0.0312	0.0120	0.0276
					RM	CL-TTR3-005	5.79	0.0261	0.0377	0.318	34.8	0.185	0.0249	0.0321
				Background	CT	CL-TTR8-002	0.0540	0.0660	0.0484	1.28	1.20	0.0580	0.0336	0.0414
					RM	CL-TTR7-001	0.132	0.0403	0.148	1.38	1.69	0.0312	0.0290	0.0328
Gulf killifish	1	0.495	NA	Site	CT	GK-TTR5-2	0.201	0.0123	0.0119	0.447	0.618	0.00880	0.0112	0.0110
					RM	GK-TTR3-2	9.53	0.00995	0.00950	0.348	4.46	0.0125	0.335	0.0165
				Background	CT	GK-TTR7-2	0.120	0.0232	0.0189	0.814	0.0895	0.0218	0.0195	0.0204
					RM	GK-TTR7-1	0.169	0.0795	0.0459	0.381	0.0850	0.0405	0.0331	0.0505
Hardhead catfish	NA	0.495	NA	Site	CT	SJFCA1-LF6	23.7	0.0235	4.51	4.47	3.78	2.22	0.0198	0.0213
					RM	SJFCA1-LF1	28.1	0.0236	3.26	3.84	2.83	1.62	0.0163	0.0184
				Background	CT	SJFCACB-LF6	1.62	0.544	1.35	2.14	0.227	0.251	0.495	0.0231
					RM	SJFCACB-LF5	1.67	0.492	1.23	2.32	0.517	0.201	0.0234	0.0190
Sediment	0.02	NA	NA	Site	CT	SJB2	269	3.99	33.5	235	898	127	118	45.8
					RM	SJE1	1020	10.2	14.1	73.2	3,590	225	142	43.6
				Post-TCRA	CT	SJNE052	24.4	2.95	0.0483	0.692	0.316	9.38	0.338	0.726
					RM	SJNE052	24.4	2.95	0.0483	0.692	0.316	9.38	0.338	0.726
				Background	CT	SJUP006	0.307	0.270	8.97	64.2	1.17	0.306	0.175	3.87
					RM	SJUP015	0.117	0.106	7.90	91.7	3.40	0.0920	0.726	3.41
Shoreline sediment	NA	0.033	0.18	Site	CT	TCEQ2009_03	680	130	95.0	220	2700	145	170	75.0
					RM	SJNE022-2	1600	13.4	12.8	80.6	4930	466	371	107
				Post-TCRA	CT	SJSH002	7.65	0.788	0.0160	0.235	0.163	2.75	0.118	0.337
					RM	SJSH021	7.69	6.41	0.0385	0.273	0.0351	25.2	0.703	0.130
				Background	CT	SJSH055	0.0342	0.0268	2.86	20.2	0.826	0.183	0.0178	0.790
					RM	SJSH049	0.0182	0.237	1.25	13.3	4.38	0.702	0.443	0.650

Table 4-16
Regression Models and TEF Substitutions Used To Estimate $TEQ_{DF,B}$ Concentrations in Bird Eggs

Exposure	Regression Model Used	Min. TEF Used	Max. TEF Used
TCDD	TCDD	1	1
PeCDD	PeCDD	1	1
∑HxCDDs	HxCDD	0.01	0.1
∑HpCDD [a]	HxCDD	0.001	0.001
TCDF	TCDF	1	1
∑PeCDF	PeCDF	0.1	1
∑HxCDF [a]	PeCDF	0.1	0.1
∑HpCDF [a]	PeCDF	0.01	0.01

Sources

Regression model: Elliott et al. (2001)

TEF: Van den Berg et al. (1998)

Notes

TEF = toxicity equivalence factor

a - Regression parameters not available; parameters used were for the most closely associated homologue group.

Table 4-17

Predicted TEQ Concentrations for Each Dioxin and Furan Congener and $TEQ_{DF,B}$ in Bird Eggs for the Site

Receptor	Congener	Scenarios																			
		Prey Only				Prey + Sediment				Prey + Post-TCRA Sediment				Background: Prey Only				Background: Prey + Sediment			
		CT		RM		CT		RM		CT		RM		CT		RM		CT		RM	
		TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax	TEFmin	TEFmax
Cormorant	TCDD	7.54	7.54	216	216	136	136	584	584	9.44	9.44	230	230	4.83	4.83	6.49	6.49	5.04	5.04	6.56	6.56
	PeCDD	3.95	3.95	3.44	3.44	14.5	14.5	25.0	25.0	6.80	6.80	8.15	8.15	5.94	5.94	13.2	13.2	6.80	6.80	13.4	13.4
	∑HxCDDs	0.0304	0.304	0.0262	0.262	0.444	4.44	0.253	2.53	0.234	2.34	0.213	2.13	0.0413	0.413	0.0743	0.743	0.196	1.96	0.199	1.99
	∑HpCDD	0.0368	0.0368	0.0312	0.0312	0.182	0.182	0.0847	0.0847	0.0997	0.0997	0.0778	0.0778	0.0545	0.0545	0.0332	0.0332	0.101	0.101	0.0967	0.0967
	TCDF	1.77	1.77	3.96	3.96	7.07	7.07	12.6	12.6	1.97	1.97	4.70	4.70	0.806	0.806	0.789	0.789	0.886	0.886	1.00	1.00
	∑PeCDF	0.0753	0.753	0.0977	0.977	5.02	50.2	7.67	76.72	0.143	1.43	0.545	5.45	0.147	1.47	0.233	2.33	0.177	1.77	0.241	2.41
	∑HxCDF	0.0897	0.0897	1.12	1.12	4.76	4.76	5.91	5.91	0.279	0.279	1.42	1.42	0.136	0.136	0.201	0.201	0.153	0.153	0.263	0.263
	∑HpCDF	0.00886	0.00886	0.0120	0.0120	0.237	0.237	0.230	0.230	0.0297	0.0297	0.0548	0.0548	0.0140	0.0140	0.0275	0.0275	0.0449	0.0449	0.0518	0.0518
	TEQ	**13.5**	**14.5**	**225**	**226**	**168**	**217**	**636**	**708**	**19.0**	**22.4**	**245**	**252**	**12.0**	**13.7**	**21.0**	**23.8**	**13.4**	**16.8**	**21.8**	**25.8**
Heron	TCDD	261	261	387	387	657	657	1240	1240	261	261	391	391	26.7	26.7	28.1	28.1	26.8	26.8	28.1	28.1
	PeCDD	5.42	5.42	6.75	6.75	175	175	41.7	41.7	6.69	6.69	8.72	8.72	29.9	29.9	30.0	30.0	29.9	29.9	30.6	30.6
	∑HxCDDs	0.977	9.77	0.791	7.91	1.74	17.4	0.921	9.21	1.03	10.3	0.851	8.51	0.444	4.44	0.423	4.23	0.484	4.84	0.441	4.41
	∑HpCDD	0.112	0.112	0.101	0.101	0.276	0.276	0.173	0.173	0.137	0.137	0.130	0.130	0.0802	0.0802	0.0756	0.0756	0.0944	0.0944	0.0837	0.0837
	TCDF	2.99	2.99	3.67	3.67	13.5	13.5	17.3	17.3	3.03	3.03	3.99	3.99	1.02	1.02	1.32	1.32	1.09	1.09	1.55	1.55
	∑PeCDF	2.72	27.2	2.20	22.0	9.35	93.5	19.8	198	2.76	27.6	2.30	23.0	0.573	5.73	0.524	5.24	0.592	5.92	0.596	5.96
	∑HxCDF	0.118	0.118	0.695	0.695	9.03	9.03	16.2	16.2	0.239	0.239	0.824	0.824	0.912	0.912	0.179	0.179	0.913	0.913	0.244	0.244
	∑HpCDF	0.0118	0.0118	0.0125	0.0125	0.494	0.494	0.642	0.642	0.0360	0.0360	0.0212	0.0212	0.0147	0.0147	0.0207	0.0207	0.0264	0.0264	0.0297	0.0297
	TEQ	**273**	**306**	**402**	**429**	**867**	**966**	**1340**	**1530**	**275**	**309**	**408**	**437**	**59.6**	**68.8**	**60.6**	**69.2**	**59.9**	**69.6**	**61.6**	**70.9**
Sandpiper	TCDD	45.6	45.6	114	114	2010	2010	4240	4240	49.1	49.1	139	139	2.66	2.66	5.10	5.10	2.89	2.89	5.21	5.21
	PeCDD	18.3	18.3	48.4	48.4	524	524	138	138	21.9	21.9	52.3	52.3	9.99	9.99	7.94	7.94	10.6	10.6	13.1	13.1
	∑HxCDDs	0.300	3.00	0.515	5.15	3.80	38.0	1.22	12.2	0.719	7.19	0.873	8.73	0.207	2.07	0.260	2.60	0.464	4.64	0.375	3.75
	∑HpCDD	0.0400	0.0400	0.0442	0.0442	0.700	0.700	0.369	0.369	0.185	0.185	0.200	0.200	0.0606	0.0606	0.0583	0.0583	0.157	0.157	0.126	0.126
	TCDF	4.69	4.69	7.65	7.65	26.8	26.8	34.5	34.5	4.83	4.83	8.24	8.24	1.90	1.90	2.13	2.13	2.05	2.05	2.71	2.71
	∑PeCDF	1.21	12.1	3.22	32.2	28.5	285	67.7	677	1.49	14.9	3.71	37.1	0.413	4.13	0.303	3.03	0.523	5.23	0.716	7.16
	∑HxCDF	0.271	0.271	0.591	0.591	31.7	31.7	56.6	56.6	0.732	0.732	1.25	1.25	0.168	0.168	0.198	0.198	0.184	0.184	0.497	0.497
	∑HpCDF	0.0158	0.0158	0.0172	0.0172	1.73	1.73	2.25	2.25	0.111	0.111	0.0541	0.0541	0.0194	0.0194	0.0159	0.0159	0.0687	0.0687	0.0588	0.0588
	TEQ	**70.4**	**84.0**	**175**	**208**	**2630**	**2920**	**4540**	**5160**	**79.0**	**98.9**	**205**	**247**	**15.4**	**21.0**	**16.0**	**21.1**	**16.9**	**25.8**	**22.8**	**32.7**

Notes

CT = central tendency

RM = reasonable maximum

TEF = toxicity equivalence factor

TEQ = toxicity equivalent (ng/kg)

Table 4-18
Fish-to-Egg Biomagnification Factors for Selected PCB Congeners

PCB Congener	TEF-Bird (WHO 1998)	Detection Frequency in Onsite Sediments	Correlates with TCDD and TCDF in Onsite Sediments?	Herring Gull BMF[a]	Gray Heron BMF[b]	Kingfisher BMF[b]
Assessment Species - Background				**Cormorant**	**Blue Heron**	**Sandpiper**
Non-ortho Substituted PCBs						
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.05	15/27 (56%)	Y	18.1	0.7	0.16
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.1	5/27 (19%)	N	18.1	14.8	3.45
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	3/27 (11%)	N	18.7	20.4	4.74
Mono-ortho Substituted PCBs						
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.0001	23/27 (85%)	Y	20	17.4	4.06
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.0001	15/27 (56%)	Y	18.7	14.4	3.36
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00001	22/27 (81%)	Y	31	19.8	4.61

Notes
BMF = biomagnification factor
PCB = polyclorinated biphenyl
TEF = toxicity equivalence factor
a - Braune and Norstrom (1989). These authors present fish tissue (alewife) and egg data (herring gulls) for several congeners, but among dioxin-like PCB congeners, only two are represented: PCB 105 and PCB 118. BMFs shown for those not represented are an average for the relevant homologue group.
b - Naito and Murata (2007)

Table 4-19

Estimated $TEQ_{P,B}$ (ng/kg wet weight) for Selected PCB Congeners in Bird Eggs for Each Exposure Scenario [a]

		$TEQ_{P,B}$ (ng/kg wet weight)									
		Prey Only		Prey + Sediment		Prey + Post-TCRA Sediment		Background: Prey		Background: Prey + Sediment	
Receptor	**Congener**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cormorant	PCB077	29.6	20.8	30.4	25.1	29.7	21.1	5.11	4.00	5.18	4.29
	PCB081	1.14	2.21	1.21	2.31	1.26	2.24	1.76	1.96	1.88	2.03
	PCB105	1.43	5.70	1.64	6.01	1.43	5.72	0.440	0.436	0.441	0.439
	PCB114	0.0819	0.380	0.0915	0.395	0.0820	0.381	0.0290	0.0316	0.0291	0.0317
	PCB118	0.654	2.27	0.733	2.39	0.655	2.28	0.216	0.213	0.216	0.214
	PCB126	12.8	36.5	13.3	36.8	13.0	36.6	2.50	10.2	2.64	10.2
	TEQ	**45.7**	**67.8**	**47.4**	**73.0**	**46.1**	**68.3**	**10.1**	**16.8**	**10.4**	**17.2**
Heron	PCB077	2.00	1.54	2.28	2.03	NA	NA	1.01	0.655	NA	NA
	PCB081	6.66	6.02	6.79	7.02	NA	NA	2.89	3.03	NA	NA
	PCB105	16.2	15.2	16.6	15.8	NA	NA	4.97	3.40	NA	NA
	PCB114	0.802	0.767	0.821	0.794	NA	NA	0.262	0.181	NA	NA
	PCB118	3.45	5.13	3.58	5.31	NA	NA	2.23	1.54	NA	NA
	PCB126	46.9	93.3	47.4	95.3	NA	NA	11.0	30.4	NA	NA
	TEQ	**75.9**	**122**	**77.4**	**126**	NA	NA	**22.4**	**39.2**	NA	NA
Sandpiper	PCB077	0.498	0.797	0.841	1.41	NA	NA	0.102	0.128	NA	NA
	PCB081	0.879	0.708	1.05	1.98	NA	NA	0.281	0.295	NA	NA
	PCB105	0.365	0.272	0.930	1.01	NA	NA	0.0329	0.0609	NA	NA
	PCB114	0.0199	0.0117	0.0443	0.0472	NA	NA	0.00195	0.00346	NA	NA
	PCB118	0.124	0.0902	0.282	0.312	NA	NA	0.0206	0.0224	NA	NA
	PCB126	2.37	1.94	3.04	4.40	NA	NA	0.429	0.508	NA	NA
	TEQ	**4.26**	**3.82**	**6.19**	**9.15**	NA	NA	**0.867**	**1.02**	NA	NA

Notes

CT = central tendency

RM = reasonable maximum

TEF = toxicity equivalence factor

a - Not all PCB congeners are represented because biomagnification factors for a full suite of dioxin-like PCB congeners are not presented by any one study, nor for any one species. Selected congeners are those with relatively high TEFs, or which were commonly detected in Site sediments (Table 4-18).

Table 4-20

Estimated Concentrations of $TEQ_{DF.B}$, $TEQ_{P.B}$, and $TEQ_{DFB.P}$ in Bird Eggs under Each Exposure Scenario [a]

Receptor	Scenario	Max $TEQ_{DF,B}$ (ng/kg wet wt)		$TEQ_{P,B}$ (ng/kg wet wt)		$TEQ_{DFP,B}$ (ng/kg wet wt)	
		CT	RM	CT	RM	CT	RM
Cormorant	prey (Gulf killifish)	14.5 (24.0%)	226 (76.9%)	45.7 (76.0%)	67.8 (23.1%)	60.2	294
	prey + sediment	217 (82.1%)	708 (90.7%)	47.4 (17.9%)	73.0 (9.74%)	265	781
	prey + post-TCRA sediment	22.4 (32.7%)	252 (78.7%)	46.1 (67.3%)	68.3 (21.3%)	68.5	320
	prey - background	13.7 (57.6%)	23.8 (58.6%)	10.1 (42.4%)	16.8 (41.4%)	23.7	40.6
	background prey + sediment	16.8 (61.8%)	25.8 (60%)	10.4 (38.2%)	17.2 (40.3%)	27.1	43.0
Great blue heron	prey (Gulf killifish, blue crab, hardhead catfish)	306 (80.1%)	429 (77.8%)	75.9 (19.9%)	122 (22.2%)	382	551
	prey + sediment	966 (92.6%)	1,530 (92.4%)	77.4 (7.42%)	126 (7.63%)	1,040	1,650
	prey + post-TCRA sediment	309 (100%)	437 (100%)	- [b]	- [b]	309	437
	prey - background	68.8 (75.5%)	69.2 (63.8%)	22.4 (24.5%)	39.2 (36.2%)	91.2	108
	background prey + sediment	69.6 (100%)	70.9 (100%)	- [b]	- [b]	69.6	70.9
Sandpiper	prey (common rangia, blue crab)	84.0 (95.2%)	208 (98.2%)	4.26 (4.83%)	3.82 (1.80%)	88.2	212
	prey + sediment	2,920 (99.8%)	5,160 (99.8%)	6.19 (0.211%)	9.15 (0.177%)	2,920	5,170
	prey + post-TCRA sediment	98.9 (100%)	247 (100%)	-	-	98.9	247
	prey - background	21.0 (96.0%)	21.1 (95.4%)	0.870 (3.98%)	1.02 (4.62%)	21.9	22.1
	prey - background	25.8 (100%)	32.7 (100%)	- [b]	- [b]	25.8	32.7

Notes

CT = central tendency

RM = reasonable maximum

TEQ = toxicity equivalent (ng/kg)

a - Percent contribution to $TEQ_{DFP,B}$ is shown.

b - There are no PCB congener data in upstream shoreline sediments

Table 4-21

Parameter Distributions Used for Probabilistic Exposure and Risk Assessment for Wildlife Receptors[a]

Receptor	Distribution Type	Central Tendency	SD	Range	Reference
Sandpiper					
Body Weight (kg)	Normal	0.0471	0.0018	0.043-0.050	DREBWQAT (1999), Maxson and Oring (1980)
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.18	NA	0.073-0.30	Beyer et al. (1994); mean and range for four sandpiper species
Diet – Crabs (Fraction of Diet)	Uniform	NA	NA	0-1	BPJ
Diet – Clams (Fraction of Diet)	Uniform	NA	NA	0-1	BPJ; fraction in diet for clams calculated in each iteration after random selection of fraction in diet for crabs
Killdeer					
Body Weight (kg)	Normal	0.101	0.0037	0.0922-0.107	Jackson and Jackson (2000) for CT of adult female; range and SD based on scaling sandpiper data to killdeer CT
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.10	NA	0.02-0.2	Beyer et al. (1994) for CT; BPJ for range
Diet – Terrestrial Invertebrates (Fraction of Diet)	Triangular	0.98	NA	0.5-0.99	Jackson and Jackson (2000) for CT; BPJ for range
Diet – Plants (Fraction of Diet)	Triangular	0.02	NA	0.01-0.5	Jackson and Jackson (2000) for CT; BPJ for range; fraction in diet for plants calculated in each iteration after random selection of fraction in diet for terrestrial invertebrates
Marsh Rice Rat					
Body Weight (kg)	Normal	0.0677	0.0134[b]	N/A	Fernandes (2011)
Sediment Ingestion Rate (Fraction of Diet)	Triangular	0.02	NA	0.01-0.1	Beyer et al. (1994) for CT based on <0.02 for white-footed mouse, and for range based on BPJ and values for black-tailed prairie dog, opossum, and raccoon

Table 4-21

Parameter Distributions Used for Probabilistic Exposure and Risk Assessment for Wildlife Receptors[a]

Receptor	Distribution Type	Central Tendency	SD	Range	Reference
Diet – Crabs (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]
Diet – Clams (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]
Diet – Plants (Fraction of Diet)	Triangular	0.4	NA	0.132-1	Wolfe (1982) for CT; BPJ for range[c]
Diet - Small Fish (Fraction of Diet)	Triangular	0.2	NA	0.066-0.6	Wolfe (1982) for CT; BPJ for range[c]

Notes

NA = not applicable

BPJ = best professional judgment

SD = standard deviation

a - Feeding rate and water ingestion rate were calculated from body weight value using allometric equations in each iteration of the Monte Carlo analysis. Home range was not used in the exposure model for these receptors because Area Use Factor was assumed equal to 1.0.

b - Standard deviation was calculated from the supplied standard error and population sample size provided in Fernandes (2011).

c - For each iteration of the Monte Carlo analysis, values for the dietary components were randomly selected from the specified distributions and then normalized so that all components summed to 1.0. The normalization process included dividing each dietary component value by the sum total of the dietary component values.

Table 5-1
Toxicity Reference Values and Benchmarks for Benthic Macroinvertebrates

Chemical	Sediment Concentration (ng/kg dw for organics; mg/kg dw for metals) TRV Type	Value	Ref	Water Concentration[a] (µg/L) TRV Type	Value	Ref	Endpoint/Comments
Organic Compounds							
2,3,7,8-TCDD	NOAEC	2,343		NA	NA		Geometric mean of NOAECs for a range of invertebrate taxa from Table B-4
Bis(2-ethylhexyl)phthalate	--	ND		NOAEC[b]	100	c	Opossum shrimp and amphipod mortality in 4 day lab test. NOAEC is LC_{50} ÷10.
Carbazole	--	ND					No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Phenol	--	ND		NOAEC[b]	26	d	Mysid shrimp mortality in 4 day lab test. NOAEC is LC_{50} ÷ 10.
Metals							
Aluminum	--	ND		NOAEC[b]	1,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Barium	--	ND		--	ND		No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Cobalt	--	ND		NOAEC[b]	450	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Copper	ER-L	34	f	--			
	ER-M	270	f	AWQC (CCC)	3.1	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]
Lead	ER-L	46.7	f	--			
	ER-M	218	f				
Manganese	--	ND		NOAEC[b]	7,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Mercury	ER-L	0.15	f	--			
	ER-M	0.71	f	AWQC (CCC)	0.94	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]
Thallium	--	ND		NOAEC[b]	213	h	Derived from acute toxicity to marine life . NOAEC is EC ÷ 10. Details unavailable.
Vanadium	--	ND		NOAEC	5	i	NOAEC is EC_{50} ÷10 in most sensitive species. Effect is development.
				LOAEC	10	i	LOAEC is EC_{50} ÷ 10 in most sensitive species. Effect is development.
Zinc	ER-L	150	f	--			
	ER-M	410	f	AWQC (CCC)	81	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected. [g]

Notes

-- = Risks were not evaluated using lines of evidence requiring this information.
AWQC = Ambient Water Quality Criteria. Criterion Continuous Concentrations shown
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
EC = effects concentration
ER-L = effect range-low: concentration below which effects are rarely observed or predicted among sensitive life stages and (or) species of biota
ER-M = effect range-median: concentration above which effects are frequently or always observed among most species of biota
USEPA = U.S. Environmental Protection Agency
WHO = World Health Organization

a - TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - TRV is an LC_{50} divided by an uncertainty factor of 10.
c - Ho et al. (1997)
d - Kim and Chin (1995)
e - Bengtsson (1978)
f - Long et al. (1995)
g - Ambient Water Quality Criteria Website (http://water.epa.gov/scitech/swguidance/standards/current/index.cfm#altable)
h - USEPA (1986)
i - WHO (2001)

Table 5-2
Toxicity Reference Values and Benchmarks for Fish

Chemical	Water Concentration[a] (µg/L)		Ref	Fish Food[b] (mg/kg dw)		Ref	Fish Whole Body		Units	Ref	Comments
Organic Compounds											
TCDD (mg/kg lipid)	--	--		--	--		NOAEL	0.321	µg/kg lipid	c	From a species sensitivity distribution; protects 95 percent of fish species. Endpoint is egg survival.
PCBs	--	--		--	--		NOAEL	5.0	mg/kg ww	d	Geometric mean of NOAELs from 3 fish species.
	--	--		--	--		LOAEL	16	mg/kg ww	d	Geometric mean of LOAELs across 3 fish species.
Bis(2-ethylhexyl)phthalate	NOAEL	55,000	e	--	--		--	--	--		Derived from 4-day acute test with sheepshead minnow. NOAEL is LC_{50} ÷ 10. Endpoint is survival.
Metals											
Cadmium				LOAEL	14.1	f	--	--	--		
Copper				NOAEL	50	g	--	--	--		
				LOAEL	100	h	--	--	--		
Mercury				NOAEL	0.5	i	--	--	--		Endpoint is F_0 male survival in mummichog resulting from increased aggression due to neurotoxic effects. aquarium confinement, or both.
				LOAEL	1.9	i	--	--	--		
Nickel	NOAEL	3,600	j, k	ND			--	--	--		Geometric mean of NOECs for several marine fish. See Table B-17 and Appendix B text.
Zinc				NOAEL	1,900	l	--	--	--		Fish exposed to multiple metals in water as well as food. Fish fed live Artemia exposed to zinc chloride in water. Endpoints are growth and survival.
				LOAEL	2,000		--	--	--		Fish fed at same dose of zinc with 0.5% calcium experienced no adverse effects. Endpoint is growth.

Notes

AWQC = ambient water quality criteria
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
TRV = toxicity reference value
-- = Risks were not evaluated using lines of evidence requiring this information.

a - Includes AWQC and TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - Windward (2011). Values presented are lowest NOAEC with a bounded LOAEC.
c - Steevens et al. (2005)
d - See Table B-11
e - TRV is an LC_{50} divided by an uncertainty factor of 10
f - Hatakayama and Yasuo (1987), as cited in Windward (2011b)
g - Windward (2011b)
h - Windward (2011b)
i - Matta et al. (2001)
j - Hunt et al. (2002)
k - USEPA (1988) Ambient Water Quality Criteria Document for Nickel
l - Windward (2007)

Table 5-3
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	2	a	Reproduction	Geometric mean of NOAELs for 5 bird species (Table B-11). See Appendix B.
	LOAEL	3			Geometric mean of LOAELs for 4 bird species (Table B-11). See Appendix B.
TCDD (ingested dose)	NOAEL ng/kg-d	14	b	Hen mortality and egg mortality	Ingested dose was estimated from weekly injected dose.
	LOAEL ng/kg-d	140			
TCDD (egg concentration ng/kg ww)	NOAEL	450	c	Egg mortality	Derived from multiple studies. See Appendix B
	LOAEL	2,400			
Bis(2-ethylhexyl)phthalate	NOAEL	74.9	d	Growth	Unbounded NOAEL for body weight
	LOAEL	--			
Metals					
Cadmium	NOAEL	1.47	f	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.37		Reproduction	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Chromium	NOAEL	2.66	g	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	2.78			Minimum bounded LOAEL for a mortality/growth/repro endpoint
Copper	NOAEL	4.05	h	Reproduction, growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	12.1			
Lead	NOAEL	1.63	i	Reproduction	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	1.94			Lowest bounded LOAEL
Mercury	NOAEL	0.078	j	Reproduction	One dose only tested. Unbounded NOAEL for first generation.
	LOAEL	0.9	k	Reproduction	Administered as methylmercury.

Table 5-3
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Nickel	NOAEL	6.71	l	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	11.5		Growth	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Thallium	NOAEL	0.35	m	Survival	This is an LC50 multiplied by an uncertainty factor of 0.01. No LOAEC was available
Vanadium	NOAEL	0.344	n	Growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	0.413		Reproduction	Lowest bounded LOAEL for survival, growth, or reproduction
Zinc	NOAEL	66.1	o	Reproduction	Geomean of NOAELs for reproduction and growth
	LOAEL	86.6			Lowest bounded LOAEL for survival, growth, or reproduction

Notes

EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NA = not available
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Risebrough and Anderson (1975)
b - Nosek et al. (1992a)
c - Appendix B
d - O'Shea and Stafford (1980)
e - Johnson et al. (1960)
f - EcoSSL (USEPA 2005b)
g - EcoSSL for Cr(III) (USEPA 2008)
h - EcoSSL (USEPA 2007d)
i - EcoSSL (USEPA 2005c)
j - Heinz (1979)
k -Hill and Schaffner (1976)
l - EcoSSL (USEPA 2007e)
m - USEPA (1999)
n - EcoSSL (USEPA 2005d)
o - USEPA (2007f)

Table 5-4
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	0.98	a	Reproduction	Geometric means of NOAELs and LOAELs from toxicity studies with mice. See Appendix B.
	LOAEL	2			
TCDD	NOAEL	0.000001	b	Reproduction	Converted from dietary concentration to dose using assumed body weight and consumption rate.
	LOAEL	0.00001			
Bis(2-ethylhexyl)phthalate	NOAEL	5.8	c	Reproduction	Effects seen at 29 and 147 mg/kg/day doses might be age-related, in which case NOAEL and LOAEL would be under-estimated
	LOAEL	29			
Metals					
Cadmium	NOAEL	2	d	Geometric mean of bounded NOAELs for growth, mortality, repro	38 bounded NOAELs/LOAELs included in calculation
	LOAEL	10		Geometric mean of associated LOAELs	
Chromium	NOAEL	2.40	e	Reproduction, growth	Geomean of NOAELs for reproduction and growth
	LOAEL	2.82		Mortality	No unbounded LOAELs. This is the minimum unbounded LOAEL for a mortality/growth/repro endpoint.
Copper	NOAEL	5.6	f	Reproduction, growth, survival	Highest bounded NOAEL beneath the lowest bounded LOAEL
	LOAEL	9.34			
Lead	NOAEL	4.7	g	Survival	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	5.0		Growth	Lowest bounded LOAEL
Mercury	NOAEL	0.015	h	Survival and growth	Converted from dietary concentration to dose using assumed body weight and consumption rate. Converted to chronic from subchronic exposure period. Administered as methylmercury chloride.
	LOAEL	0.025			
Nickel	NOAEL	1.7	i	Reproduction	Highest bounded NOAEL below the lowest bounded LOAEL for a mortality/growth/repro endpoint
	LOAEL	2.71			Minimum bounded LOAEL for a mortality/growth/repro endpoint

Table 5-4
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Thallium	NOAEL	0.071	j	Reproduction	No NOAEL was provided. This NOAEL is the LOAEL multiplied by 0.1. Rats were exposed in drinking water. TRV may overstate bioavailability.
	LOAEL	0.71			
Zinc	NOAEL	75.4	k	Reproduction	Geomean of NOAELs for reproduction and growth; lowest bounded LOAEL for survival, reproduction and growth
	LOAEL	75.9			

Notes

Eco-SSL = Interim Eco-SSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Aulerich and Ringer (1977)
b - Murray et al. (1979)
c - David et al. (2000)
d - EcoSSL (USEPA 2005b)
e - EcoSSL (USEPA 2008)
f - EcoSSL (USEPA 2007d)
g - EcoSSL (USEPA 2005c)
h - Sample et al. (1996)
i - EcoSSL (USEPA 2007e)
j - Formigli et al. (1986)
k - USEPA (2007f)

Table 5-5
Summary of Egg Mortality TRVs; Maternal Transfer and Yolk Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	328	1,477	MT	a	Egg concentrations estimated on the basis of maternal dose of 1µg/kg for no effects and an estimated 50 percent egg mortality at 4.5 µg/kg bw, assuming a 1 percent maternal transfer into eggs (mean egg wt = 30.5 g) Nosek et al. (1992a; 1993).
$[Egg]_{TCDD}$	100	1,000	YI	b	Egg concentration associated with 10 percent egg mortality
GeoMean for Pheasants	181	1,215			
Double crested cormorant					
$[Egg]_{TCDD}$	1,000	4,000	YI	c	LOAEL is associated with 23.3 percent increase in egg mortality over egg mortality in vehicle controls
$[Egg]_{TCDD}$	1,300	5,400	YI	d	LOAEL is associated with 25.5 percent increase in egg mortality over egg mortality in vehicle controls
GeoMean for Cormorants	1,140	4,648			
FinalGeoMean	450	2,400			Geometric means rounded to two significant figures for use as TRVs
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	YI	e	LOAEL is associated with 100 percent egg mortality over control egg mortality
$[Egg]_{TCDD}$	80	160	YI	f	LOAEL is associated with 63.8 percent increase in egg mortality over egg mortality in vehicle controls
GeoMean for Chickens	89	220			
GeoMeanAll	260	1,100			

Notes

LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
MT = maternal transfer
NOAEC = no-observed-adverse effects concentration
TRV = toxicity reference value
YI = yolk injection

a - Nosek et al. (1992b)
b - Nosek et al. (1993)
c - Powell et al (1997a)
d - Powell et al. (1998)
e - Henschel et al. (1997a)
f - Powell et al. (1996)

Table 6-1
Summary of Results for Benthic Macroinvertebrates

Chemical	**HQ > 1[a] at one or more sediment sample locations (Figures 6-1 to 6-13)**
Semivolatile Organic Compounds	
Bis(2-ethylhexyl)phthalate	N[d]
Carbazole	N[b]
Phenol	Y[d]
Metals	
Aluminum	Y[c]
Barium	Y[c]
Cobalt	N[d]
Copper	Y
Lead	Y
Manganese	Y[d]
Mercury	Y
Thallium	N[d]
Vanadium	Y[c]
Zinc	Y

Notes

HQ = hazard quotient

N/A = not available (no TRV for this $COPC_E$ or addressed via sediment comparison)

Bold values are HQs ≥1

a - Individual sediment samples compared to a sediment TRV, unless otherwise noted

b - Compared to upstream maximum detection limit

c - Compared to upstream REV

d - Surface water TRV compared to estimated porewater at individual sample locations

Table 6-2
Concentrations of 2,3,7,8-TCDD in Clam Tissue (common rangia) from the Site and Background

Sample ID	2,3,7,8-TCDD (ng/kg ww)	
Site		
Transect 2 (FCA1)		
CL-TTR1-001	1.37	J
CL-TTR1-002	1.31	J
CL-TTR1-003	0.348	U
CL-TTR1-004	1.5	
CL-TTR1-005	1.42	J
Transect 3 (FCA2)		
CL-TTR3-001	***10.7***	
CL-TTR3-002	***17.6***	
CL-TTR3-003	***12.6***	
CL-TTR3-004	***13.3***	
CL-TTR3-005	**5.79**	
Transect 4 (FCA2)		
CL-TTR4-001	0.93	U
CL-TTR4-002	1.98	
CL-TTR4-003	1.64	
CL-TTR4-004	0.476	U
CL-TTR4-005	0.519	J
Transect 5 (FCA2)		
CL-TTR5-001	1.58	
CL-TTR5-002	1.18	J
CL-TTR5-003	***2.45***	
CL-TTR5-004	***2.33***	
CL-TTR5-005	1.89	
Transect 6 (FCA3)		
CL-TTR6-001	0.143	U
CL-TTR6-002	0.123	U
CL-TTR6-003	0.784	J
CL-TTR6-004	0.647	J
CL-TTR6-005	0.696	J

Table 6-2
Concentrations of 2,3,7,8-TCDD in Clam Tissue (common rangia) from the Site and Background

Sample ID	2,3,7,8-TCDD (ng/kg ww)	
Upstream Background		
CL-TTR7-001	0.132	U
CL-TTR7-002	0.244	U
CL-TTR7-003	0.454	J
CL-TTR7-004	0.261	U
CL-TTR7-005	0.175	U
CL-TTR8-001	0.0375	U
CL-TTR8-002	0.054	U
CL-TTR8-003	0.0481	U
CL-TTR8-004	0.0505	U
CL-TTR8-005	0.0625	U

Notes

Bold and *italicized* values are higher than the 2 ng/kg ww threshold in tissue associated with histology of reproductive tissues in individual female oysters.

J = Estimated value

U = Compound analyzed, but not detected above detection limit

Table 6-3
Hazard Quotients for Fish Exposed to $COPC_E$s in Food and Incidentally Ingested in Sediment

	Gulf Killifish - TTR1/TTR2				Gulf Killifish - TTR3				Gulf Killifish - TTR4				Gulf Killifish - TTR5				Gulf Killifish - TTR6				Gulf Killifish - Area-Wide			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1
Copper	0.5	0.6	0.2	0.3	0.7	0.8	0.3	0.4	0.5	0.6	0.3	0.3	0.5	0.6	0.3	0.3	0.6	0.6	0.3	0.3	0.5	0.6	0.3	0.3
Mercury	0.2	0.2	<0.1	<0.1	0.3	0.3	<0.1	<0.1	0.2	0.3	<0.1	<0.1	0.2	0.2	<0.1	<0.1	0.3	0.4	<0.1	<0.1	0.2	0.3	<0.1	<0.1
Zinc	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

	Black Drum				Southern Flounder			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM
Cadmium	NA	NA	<0.1	<0.1	NA	NA	<0.1	<0.1
Copper	0.6	0.7	0.3	0.4	0.4	0.4	0.2	0.2
Mercury	0.2	0.2	<0.1	<0.1	0.3	0.4	<0.1	0.1
Zinc	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
HQ = hazard quotient
NA = not available; TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RM = reasonable maximum
Bold values are HQs >1

Table 6-4
Hazard Quotients for Fish Exposed to $COPC_E$s in Surface Water under Pre- and Post-TCRA Conditions

$COPC_E$	Hazard Quotient
Bis(2-ethylhexyl)phthalate	<0.1
Nickel	<0.1

Notes
$COPC_E$ = chemical of potential ecological concern

Table 6-5
Hazard Quotients for Avian Receptors North of I-10 and Aquatic Areas

	Great Blue Heron				Neotropic Cormorant				Spotted Sandpiper				Killdeer			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.5	0.8	0.3	0.5
Copper	< 0.1	< 0.1	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1	**2**	**3**	0.7	0.8	0.3	**1**	< 0.1	0.4
Mercury	0.1	0.2	< 0.1	< 0.1	0.2	0.2	< 0.1	< 0.1	0.3	0.5	< 0.1	< 0.1	**2**	**7**	0.2	0.6
Nickel	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	0.3	0.1	0.2	< 0.1	0.1	< 0.1	< 0.1
Zinc	0.3	0.3	0.2	0.2	0.2	0.2	0.1	0.1	0.4	0.4	0.3	0.3	0.8	**2**	0.6	**1**
Bis(2-ethylhexyl)phthalate	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA	< 0.1	< 0.1	NA	NA
$TEQ_{DF, B}$ [a]	0.5	**1**	0.05	0.1	0.1	0.6	0.01	0.06	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**
$TEQ_{P, B}$ [b]	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.4	0.6	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
$TEQ_{DFP, B}$ [c]	0.6	**1**	< 0.1	0.1	0.2	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**
Total PCBs	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.3	0.7	0.2	0.5	< 0.1	< 0.1	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RM = reasonable maximum
Bold values are HQs≥1

Table 6-6
Hazard Quotients Based on Estimated Egg Concentrations for Birds Exposed to $TEQ_{DFP,B}$

		Max $TEQ_{DF,B}$ (ng/kg wet wt)				$TEQ_{P,B}$ (ng/kg wet wt)				$TEQ_{DFP,B}$ (ng/kg wet wt)			
Receptor	**Scenario**	**CT**		**RM**		**CT**		**RM**		**CT**		**RM**	
		NOAEL-based HQ	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**	**NOAEL-based HQ**	**LOAEL-based HQ**
Cormorant	prey (Gulf killifish)	< 0.1	< 0.1	0.5	< 0.1	0.1	< 0.1	0.2	< 0.1	0.1	< 0.1	0.7	0.1
	prey + sediment	0.5	< 0.1	**2**	0.3	0.1	< 0.1	0.2	< 0.1	0.6	0.1	**2**	0.3
	prey + post-TCRA sediment	< 0.1	< 0.1	0.6	0.1	0.1	< 0.1	0.2	< 0.1	0.2	< 0.1	0.7	0.1
	prey - Background	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Background prey + sediment	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Great blue heron	prey (Gulf killifish, blue crab, hardhead catfish)	0.7	0.1	**1**	0.2	0.2	< 0.1	0.3	< 0.1	0.8	0.2	**1**	0.2
	prey + sediment	**2**	0.4	**3**	0.6	0.2	< 0.1	0.3	< 0.1	**2**	0.4	**4**	0.7
	prey + post-TCRA sediment	0.7	0.1	**1**	0.2	NA	NA	NA	NA	0.7	0.1	**1**	0.2
	prey - background	0.2	< 0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	< 0.1	0.2	< 0.1
	Background prey + sediment	0.2	< 0.1	0.2	< 0.1	NA	NA	NA	NA	0.2	< .01	0.2	< .1
Sandpiper	prey (common rangia, blue crab)	0.2	< 0.1	0.5	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	0.2	< 0.1	0.5	< 0.1
	prey + sediment	**6**	**1**	**10**	**2**	< 0.1	< 0.1	< 0.1	< 0.1	**6**	**1**	**10**	**2**
	prey + post-TCRA sediment	0.2	< 0.1	0.5	0.1	NA	NA	NA	NA	0.2	< 0.1	0.5	0.1
	prey - background	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	Background prey + sediment	< 0.1	< 0.1	< 0.1	< 0.1	NA	NA	NA	NA	< 0.1	< 0.1	< 0.1	< 0.1

Notes

TEQ = toxicity equivalent

Values in bold are ≥ 1

Table 6-7

Hazard Quotients for Pre- and Post-TCRA Exposures-for Marsh Rice Rat, Spotted Sandpiper, and Killdeer when Pre-TCRA $HQ_L \geq 1$

Receptor	**$COPC_E$**	**Pre-TCRA Exposures[a]**				**Post-TCRA Exposures, Median-Based[b]**			
		NOAEL-based		**LOAEL-based**		**NOAEL-based**		**LOAEL-based**	
		CT	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Marsh rice rat	$TEQ_{DF, M}$	**6**	**20**	0.6	**2**	**1**	**5**	0.1	0.5
Spotted sandpiper	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Killdeer	$TEQ_{DF, B}$	**3**	**9**	0.3	**1**	0.8	**2**	< 0.1	0.2
	Zinc	0.8	**2**	0.6	**1**	0.8	**2**	0.6	**1**

Notes

$COPC_E$ = chemical of potential ecological concern

CT = central tendency

LOAEL = lowest-observed-adverse-effect level

NOAEL = no-observed-adverse-effect level

RM = reasonable maximum

TCRA = time-critical removal action

Bold values are HQs ≥ 1

a - Exposures based on concentrations in sediments prior to the TCRA.

b - Exposures based on estimated post-TCRA sediment or soil concentrations: median value from upstream background sediments used to replace sediment or soil samples within TCRA footprint, as appropriate.

Table 6-8
Hazard Quotients for Site and Background Exposures for Marsh Rice Rat, Spotted Sandpiper, and Killdeer when Site HQ_L ≥ 1

		Site Exposures				Background Exposures			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
Receptor	**$COPC_E$**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Marsh Rice Rat	$TEQ_{DF, M}$	**6**	**20**	0.6	**2**	0.2	0.2	< 0.1	< 0.1
	$TEQ_{P, M}$	0.3	0.7	< 0.1	< 0.1	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP, M}$	**7**	**20**	0.7	**2**	0.3	0.4	< 0.1	< 0.1
Spotted Sandpiper	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{P,B}$	0.4	0.6	< 0.1	< 0.1	0.1	0.1	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
Killdeer	$TEQ_{DF, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	$TEQ_{P,B}$	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	Zinc	0.8	**2**	0.6	**1**	0.7	**1**	0.6	0.8

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
TCRA = time-critical removal action
Bold values are HQs ≥ 1

Table 6-9
Hazard Quotients for Mammalian Receptors North of I-10 and Aquatic Areas

	Marsh Rice Rat				Raccoon			
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
$COPC_E$	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Cadmium	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
Copper	0.5	0.7	0.3	0.4	< 0.1	0.1	< 0.1	< 0.1
Mercury	**1**	**1**	0.6	0.8	0.3	0.6	0.2	0.4
Nickel	0.4	0.5	0.2	0.3	< 0.1	< 0.1	< 0.1	< 0.1
Zinc	0.2	0.3	0.2	0.3	< 0.1	0.1	< 0.1	0.1
Bis(2-ethylhexyl)phthalate	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1
$TEQ_{DF, M}$ [a]	**6**	**20**	0.6	**2**	**4**	**9**	0.4	0.9
$TEQ_{P, M}$ [b]	0.3	0.7	< 0.1	< 0.1	0.2	0.2	< 0.1	< 0.1
$TEQ_{DFP, M}$ [c]	**7**	**20**	0.7	**2**	**4**	**9**	0.4	0.9
Total PCBs	< 0.1	0.2	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
bold values are HQs≥1
a - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
b - Toxicity equivalent for dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
c - Toxicity equivalent for dioxins, furans and dioxin-like PCBs calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.

Table 6-10
Hazard Quotients for Endangered and Threatened Species when $HQ_N \geq 1$ for Surrogate Species

	Bald Eagle, breeding		Bald Eagle, nonbreeding		Brown Pelican		White-Faced Ibis	
	NOAEL-based		NOAEL-based		NOAEL-based		NOAEL-based	
$COPC_E$	CT	RM	CT	RM	CT	RM	CT	RM
Copper	NA	NA	NA	NA	NA	NA	<0.1	<0.1
$TEQ_{DF, B}$ [a]	<0.1	0.20	<0.1	<0.1	<0.1	<0.1	0.3	0.8
$TEQ_{DFP, B}$ [b]	0.1	0.2	<0.1	<0.1	<0.1	<0.1	0.3	0.9

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
Bold values are HQs≥1

Table 7-1
Exposure Point Concentrations for TEQ in Soils Based on the Central Tendency, Reasonable Minimum, and Reasonable Maximum Exposures of Killdeer

	Concentration in soil, ng/kg dw		
	CT	Rmin	RM
$TEQ_{DF,B}$	1,650	230	5,190

Notes

CT = central tendency

RM = reasonable maximum

RMin = reasonable minimum

Table 7-2
Hazard Quotients for Avian Receptors North of I-10 and Aquatic Areas for TEQ $_{DF,B}$ With and Without Bioavailability Adjustment for 2,3,7,8-TCDD

	Great Blue Heron				Neotropic Cormorant				Spotted Sandpiper				Killdeer				White-faced Ibis		Bald Eagle, Breeding		Bald Eagle, Wintering	
	NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based		NOAEL-based		NOAEL-based		NOAEL-based	
$COPC_E$	CT	RM	CT	RME	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM	CT	RM
$TEQ_{DF,B}$ [a]	0.5	**1**	0.05	0.1	0.1	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**3**	**9**	0.3	**1**	0.3	0.8	< 0.1	0.2	< 0.1	< 0.1
$TEQ_{DF,B}$ without RBA[b]	0.5	**1**	0.05	0.1	0.1	0.6	< 0.1	< 0.1	**10**	**30**	**1**	**3**	**4**	**10**	0.4	**1**	0.4	0.9	< 0.1	0.2	< 0.1	< 0.1

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
NA = not available; LOAEL-based TRV not available
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
RBA = relative bioavailability adjustment factor
RM = reasonable maximum
Bold values are HQs≥1
a - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-5).
b - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

Table 7-3
Hazard Quotients for Pre- and Post-TCRA Exposures of Spotted Sandpiper and Killdeer to TEQ $_{DF,B}$ with and without Bioavailability Adjustment for 2,3,7,8-TCDD

Receptor	COPC$_E$	Pre-TCRA Exposures[a]				Post-TCRA Exposures, Median-Based[b]			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
		CT	RM	CT	RM	CT	RM	CT	RM
Spotted Sandpiper	TEQ$_{DF,B}$[c]	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Spotted Sandpiper	TEQ$_{DF,B}$ without RBA[d]	**10**	**30**	**1**	**3**	0.8	**3**	< 0.1	0.3
Killdeer	TEQ$_{DF,B}$[c]	**3**	**9**	0.3	**1**	0.8	**2**	< 0.1	0.2
Killdeer	TEQ$_{DF,B}$ without RBA[d]	**4**	**10**	0.4	**1**	**2**	**5**	0.2	0.5

Notes

COPC$_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
TCRA = time-critical removal action
Bold values are HQs ≥ 1
a - Exposures based on concentrations in sediments prior to the TCRA.
b - Exposures based on estimated post-TCRA sediment or soil concentrations: median value from upstream background sediments used to replace sediment or soil samples within TCRA footprint, as appropriate.
c - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-7).
d - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit, without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

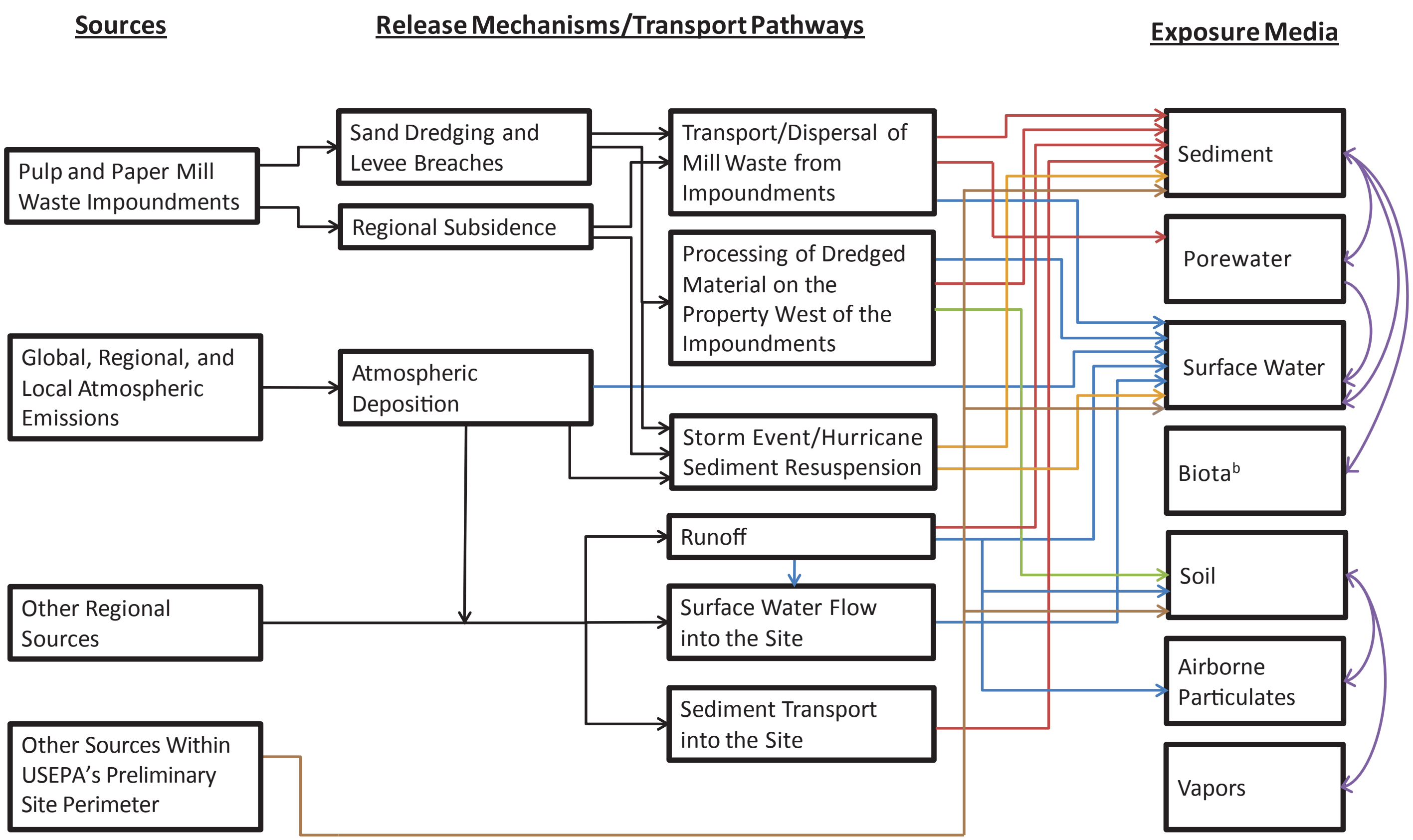
Table 7-4
Hazard Quotients for Site and Background Exposures for Spotted Sandpiper and Killdeer with and without Bioavailability Adjustment for 2,3,7,8-TCDD

		Site Exposures				Background Exposures			
		NOAEL-based		LOAEL-based		NOAEL-based		LOAEL-based	
Receptor	**$COPC_E$**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**	**CT**	**RM**
Spotted Sandpiper	RBA[a]:								
	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
	No RBA[b]:								
	$TEQ_{DF, B}$	**10**	**30**	**1**	**3**	0.1	0.2	< 0.1	< 0.1
	$TEQ_{DFP, B}$	**10**	**30**	**1**	**3**	0.2	0.3	< 0.1	< 0.1
Killdeer	RBA[a]:								
	$TEQ_{DF, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	$TEQ_{DFP, B}$	**3**	**9**	0.3	**1**	0.7	**2**	< 0.1	0.2
	No RBA[b]:								
	$TEQ_{DF, B}$	**4**	**10**	0.4	**1**	2	**5**	0.2	0.5
	$TEQ_{DFP, B}$	**4**	**10**	0.4	**1**	2	**5**	0.2	0.5

Notes

$COPC_E$ = chemical of potential ecological concern
CT = central tendency
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
RM = reasonable maximum
Bold values are HQs ≥ 1
a - Including the relative bioavailability factor adjustment for 2,3,7,8-TCDD (results as presented in Table 6-8).
b - Without the relative bioavailability factor adjustment for 2,3,7,8-TCDD.

FIGURES



Potential Receptors of Concern

Exposure Media	Human: Recreational and Subsistence Fisher	Human: Recreational Visitor	Human: Trespasser	Ecological: Benthic Macroinvertebrates[a]	Ecological: Fish	Ecological: Reptiles	Ecological: Birds	Ecological: Mammals
Sediment	●	●	○	●	●	●	●	●
Porewater	○	○	○	●	○	○	○	○
Surface Water	○	○	○	●	●	●	●	○
Biota[b]	●	x	○	●	●	●	●	●
Soil	●	●	○	x	x	●	●	●
Airborne Particulates	○	○	○	x	x	○	○	○
Vapors	○	○	○	x	x	○	○	○

● Complete, significant exposure pathway
○ Complete, minor exposure pathway
x Incomplete exposure pathway

Notes:
Other regional sources may include industrial effluents, publicly owned treatment works, and stormwater.
Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.
[a]Benthic macroinvertebrates include crabs and other crustaceans and shellfish consumed by all receptors, as well as polychaetes and other infauna consumed by fish, other marine life, birds and mammals.
[b]Biota consumed by human receptors are expected to be fish and shellfish.

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Figure 1-1
Conceptual Site Model for the Northern Impoundments and Aquatic Environment
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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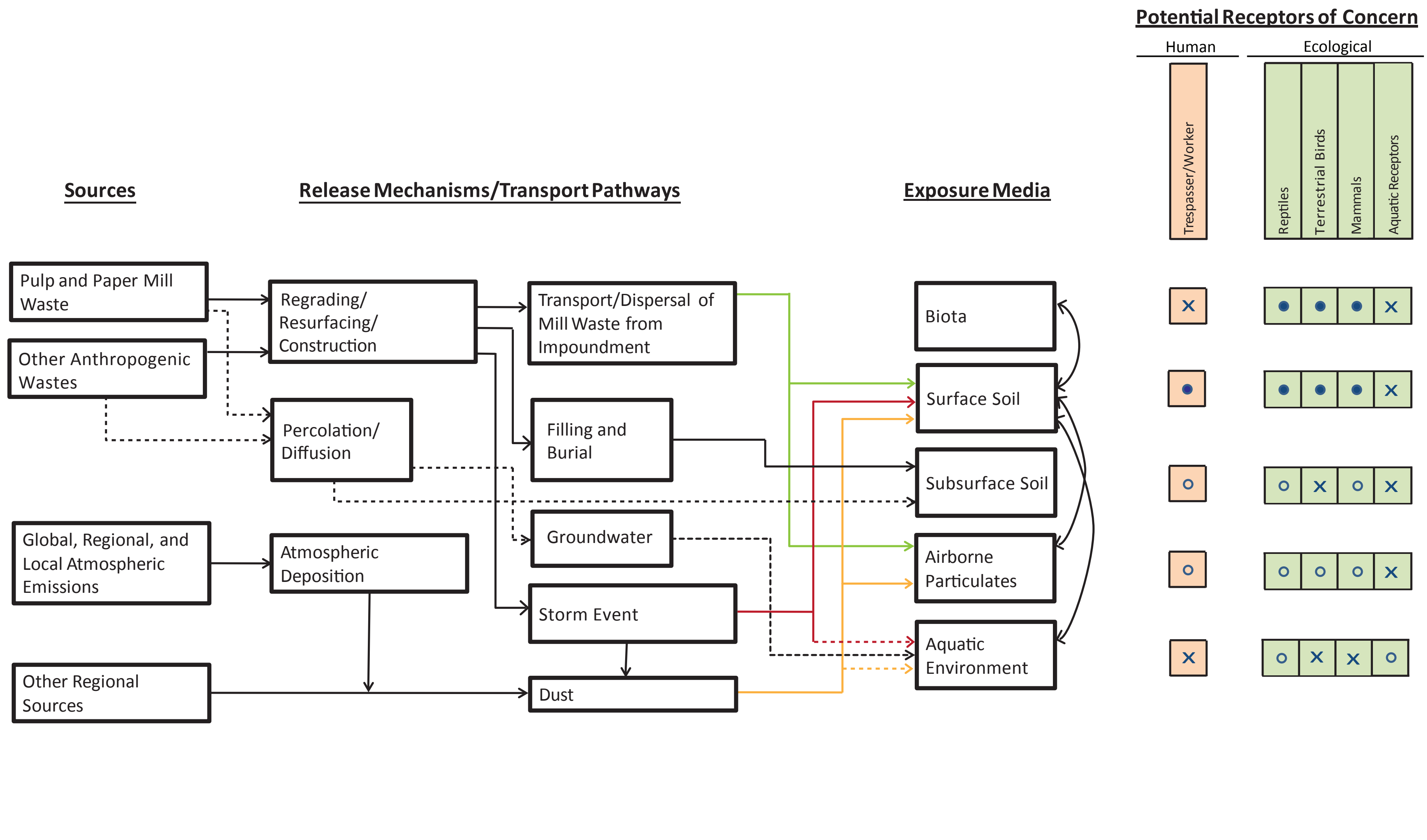
Figure 2-1
Overview of Area within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

[a] Designation of the sand separation area is intended to be a general reference to areas in which such activities are believed to have taken place based on visual observations of aerial photography from 1998 through 2002.

FEATURE SOURCES:
Aerial Imagery: 0.5-meter. Photo Date: 01/14/2009
Texas Strategic Mapping Program (StratMap), TNRIS



● Complete, significant exposure pathway
○ Complete, minor exposure pathway
x Incomplete exposure pathway
\- - - - - Unknown pathway

Notes:
Local sources may include industrial air emissions, vehicle or machinery fluid leaks, or other releases resulting from ongoing commercial activities on the site.
Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.

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Figure 2-2
Updated Conceptual Site Model Pathways for the Area South of I-10
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

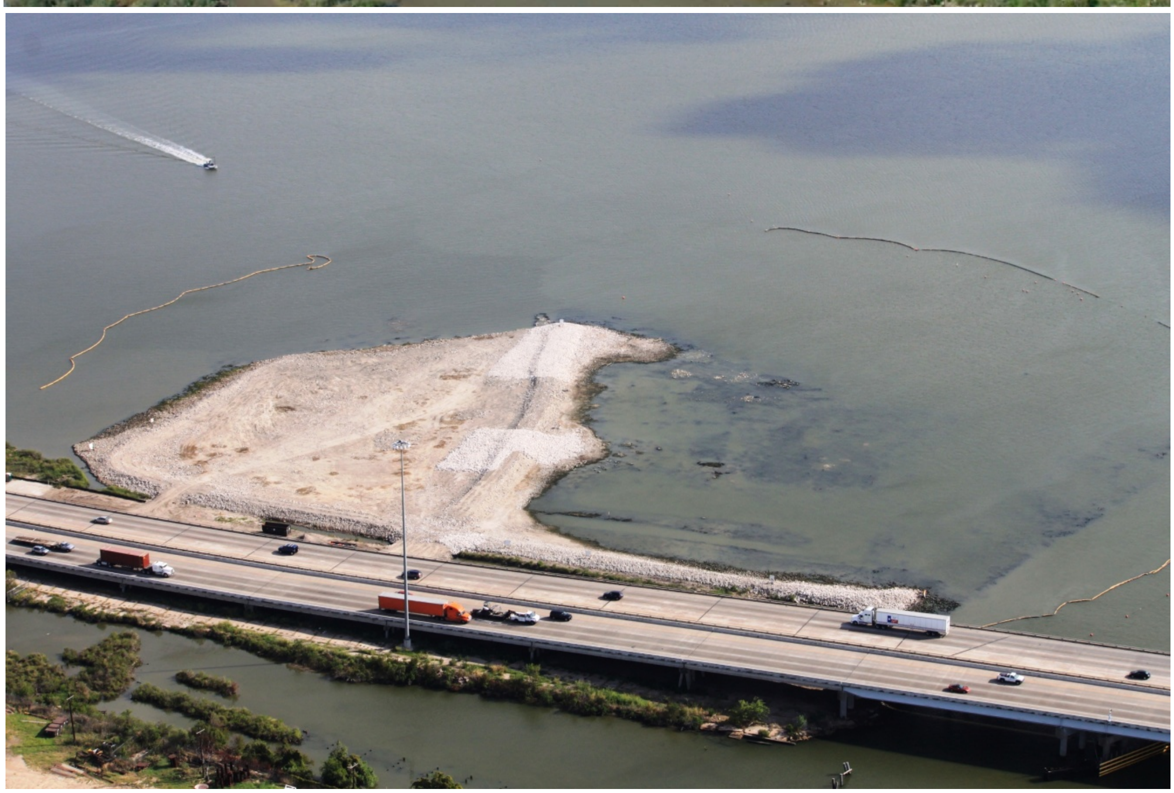
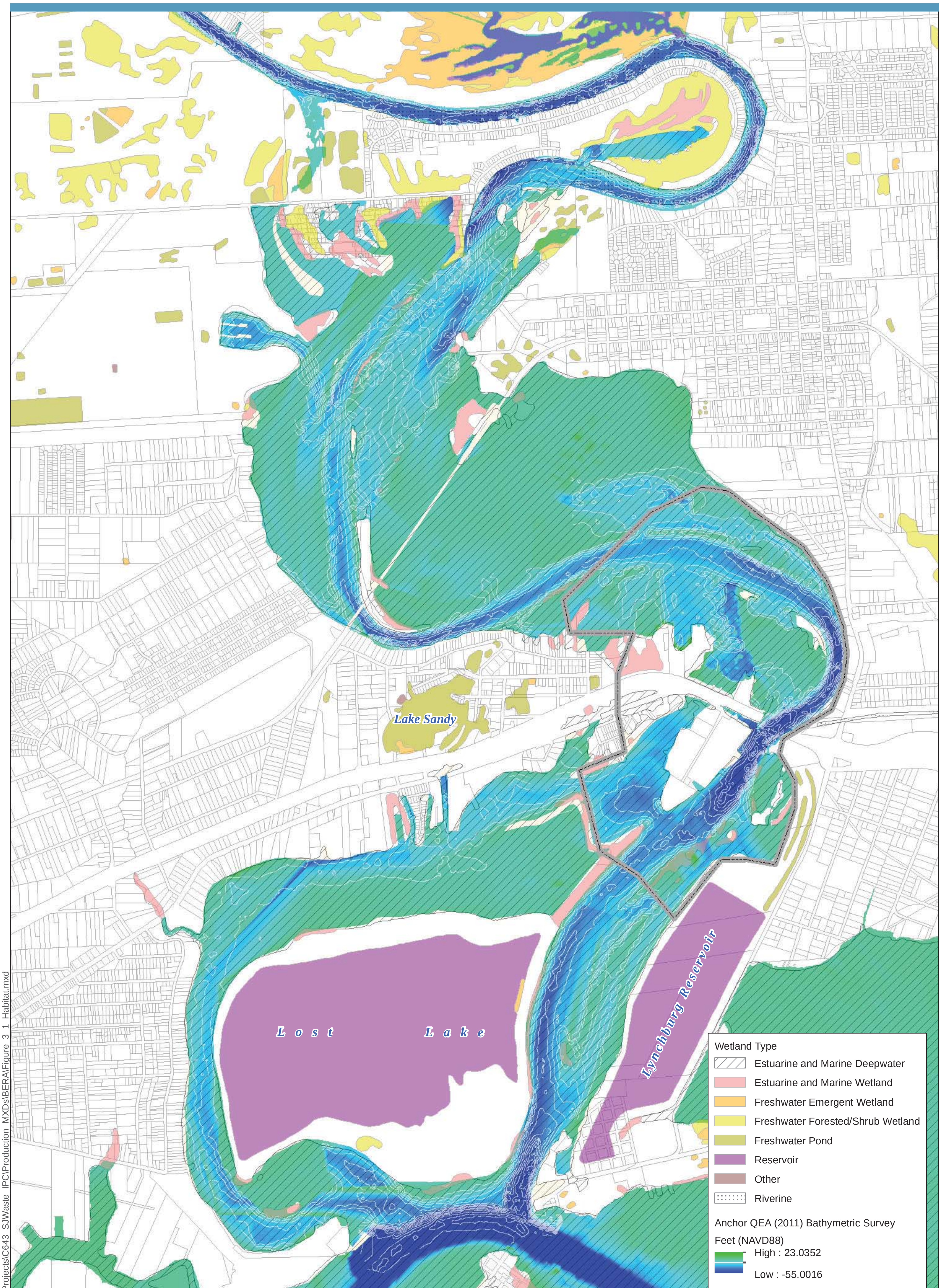


Figure 2-3
Aerial View of TCRA Project Area, Before and After
TCRA Implementation, July 14, 2011
Baseline Ecological Risk Assessment
SJRWP Superfund Site/MIMC and IPC



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USEPA's Preliminary Site Perimeter

1-Meter 1995 Bathymetric Contour

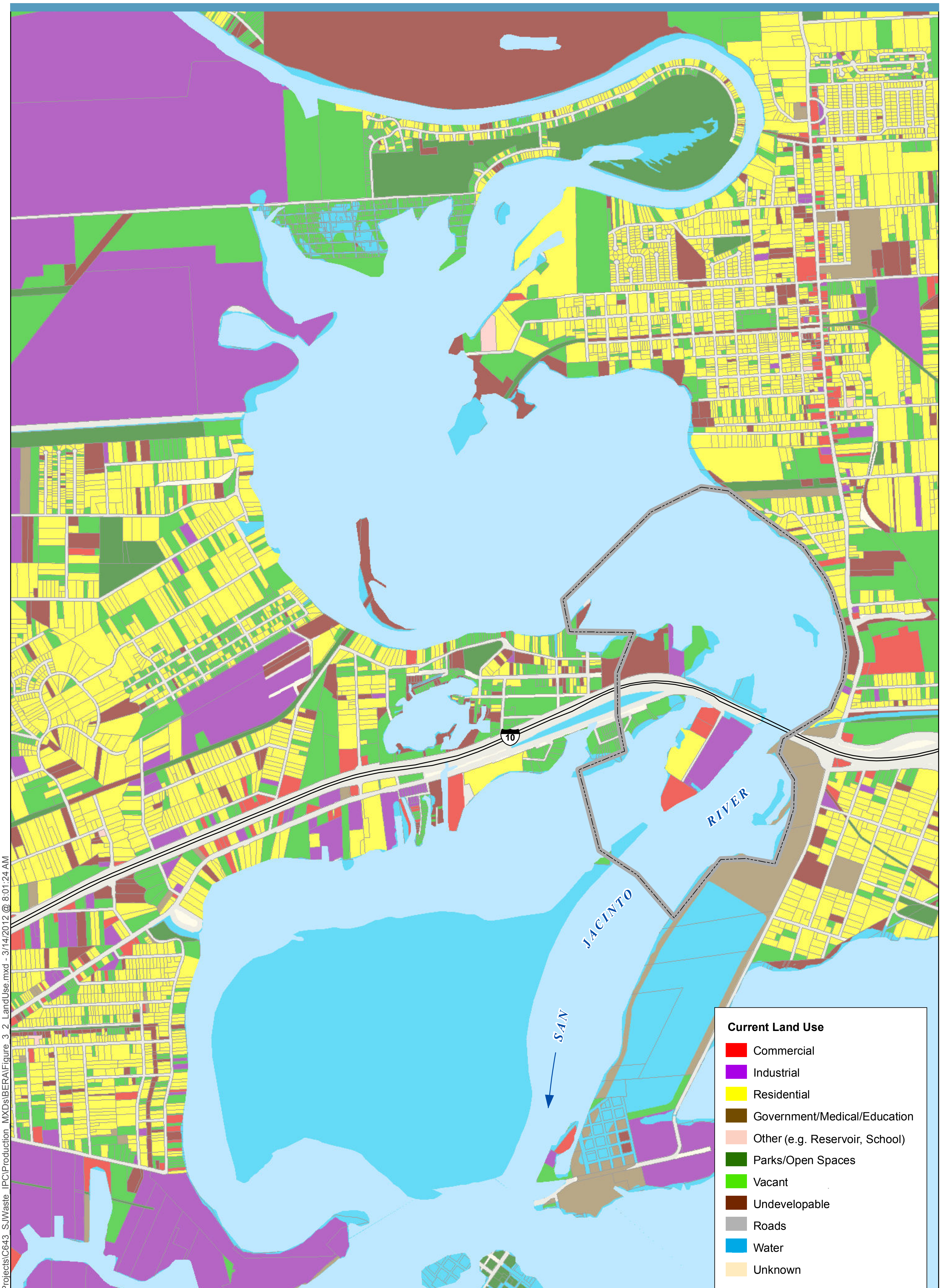
Parcel Boundary

0 0.5

Miles

FEATURE SOURCES:
Bathymetry and Contours: Anchor QEA 2011
Wetlands: U.S. Fish and Wildlife Service.
Parcel Boundaries: Harris County Appraisal District.

Figure 3-1
Habitats in the Vicinity of the Site
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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Preliminary Site Perimeter

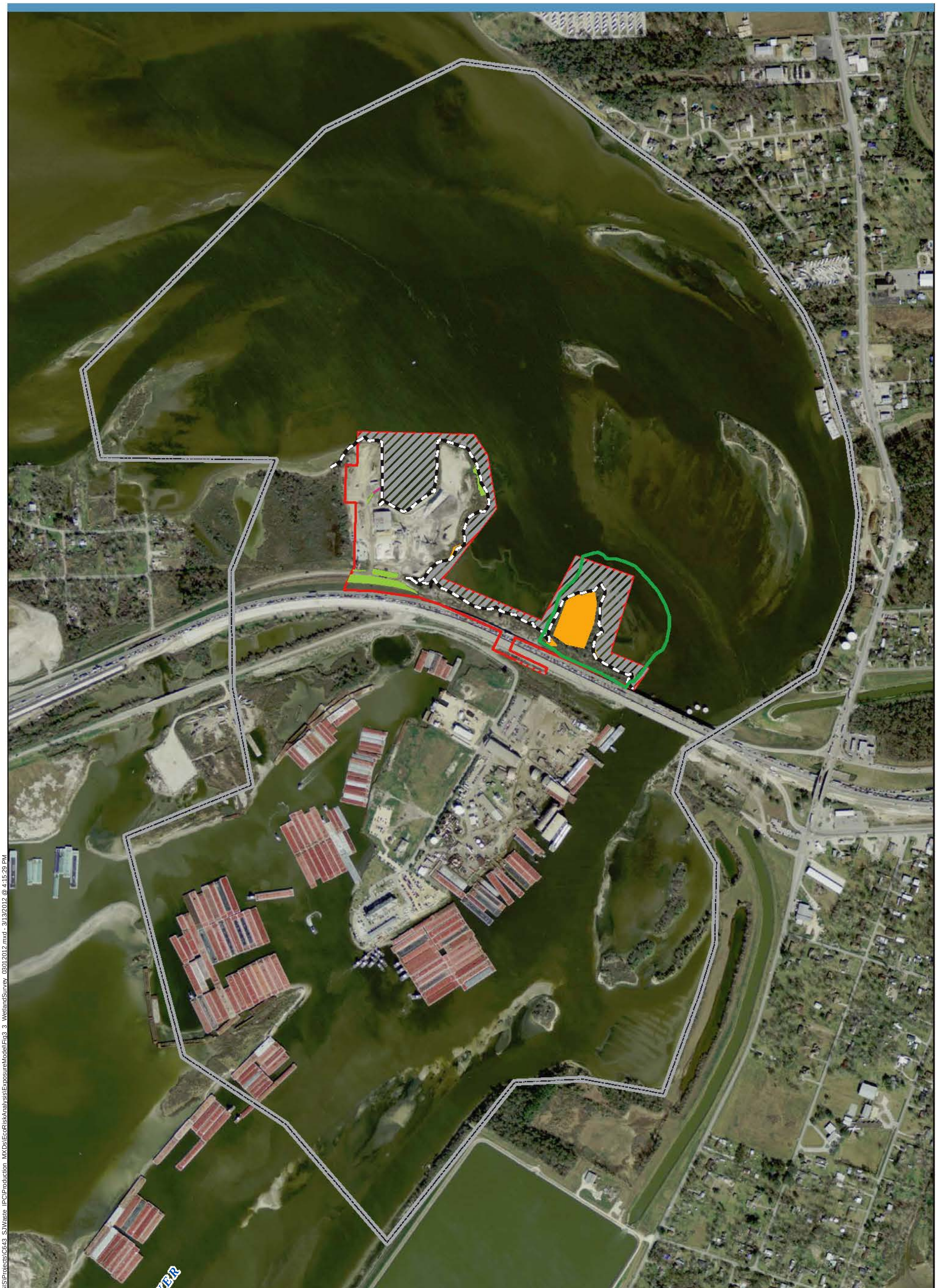
Tax Parcel Boundary

0 0.5

Scale in Miles

FEATURE SOURCES:
Zoning: Houston-Galveston Area Council
Parcel Boundaries: Harris County Appraisal District

Figure 3-2
Land Use in the Vicinity of the Site
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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0 500

Feet

Original (1966) Perimeter of the Northern Impoundments

Mean High Water

Wetland Type

Estuarine, Subtidal, Unconsolidated Bottom

Estuarine, Intertidal, Emergent

Palustrine, Emergent

Palustrine, Unconsolidated Bottom

Survey Area

USEPA's Preliminary Site Perimeter

SOURCE: BESI (2010)

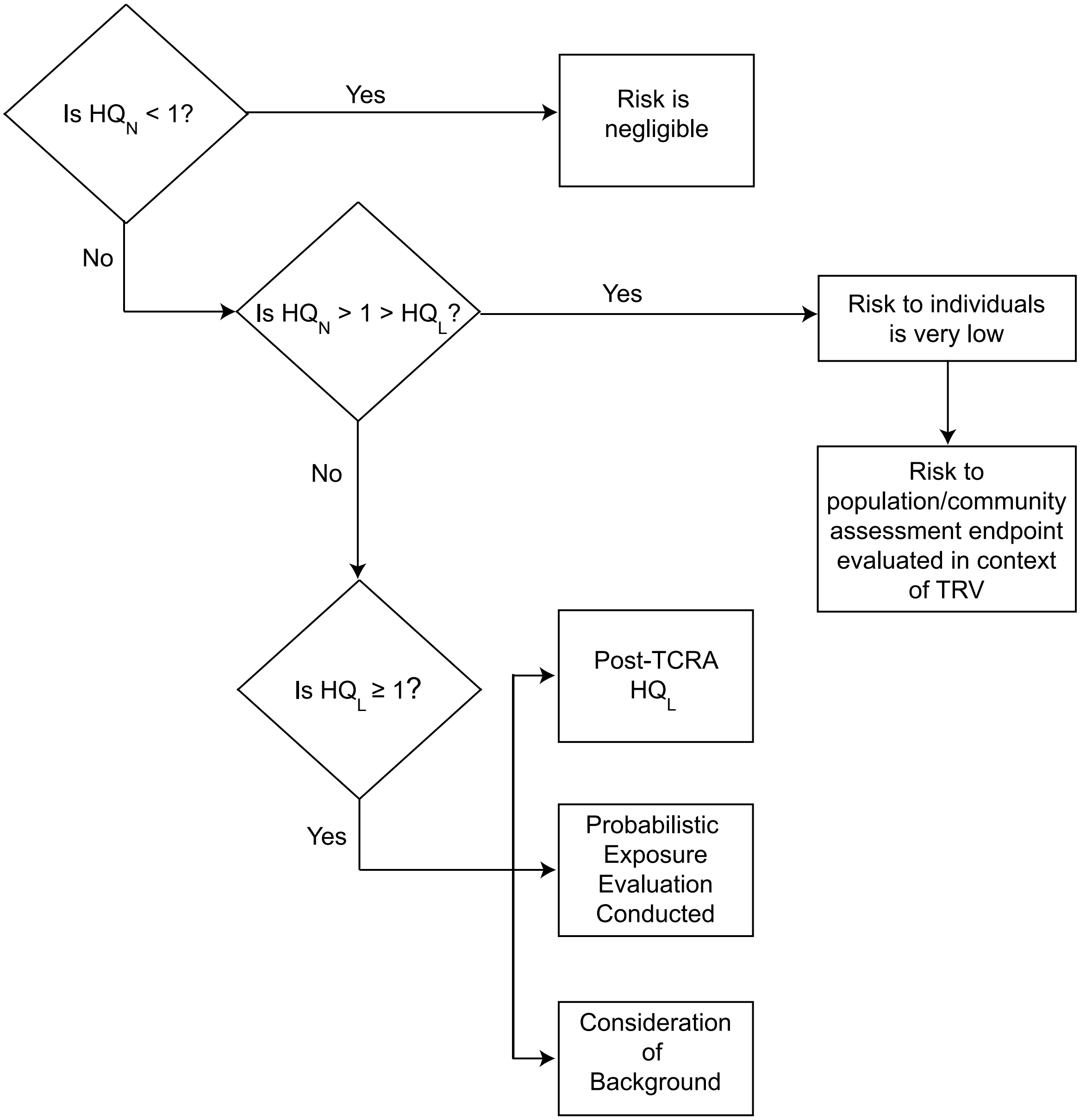
Figure 3-3

2010 Site Wetland Delineation

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC

DRAFT



Note:
HQ_L = hazard quotient based on comparison to a lowest-observed-adverse-effects level toxicity reference value.

HQ_N = hazard quotient based on comparison to a no-observed-adverse-effects level toxicity reference value.

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Figure 3-4
Decision Framework for Interpretation of Fish and Wildlife Hazard Quotients
Baseline Ecological Risk Assessment
SJRWP Superfund/IPC

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USEPA's Preliminary Site Perimeter
Original (1966) Perimeter of the Impoundments
Clams and Small Fish
Small Fish
Clams
Large Fish and Blue Crab Fish Collection Areas

0 800
Scale in Feet

[a] Designation of the sand separation area is intended to be a general reference to areas in which such activities are believed to have taken place based on visual observations of aerial photography from 1998 through 2002.

FEATURE SOURCES:
Aerial Imagery: 0.5-meter January 2009 DOQQs - Texas Strategic Mapping Program (StratMap), TNIS

Figure 4-1
Tissue Sampling Locations
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

Study Area
Clams and Small Fish
Potential Blue Crab Collection Area*

0 800
Scale in Feet

FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009 DOQQs - Texas Strategic Mapping Program (StratMap) TNRIS;

*Final collection of background blue crab tissues are to be determined in consultation with EPA, and established in an addendum to this SAP.

Figure 4-2
Upstream Background Tissue Sampling Locations
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

Blue Crab and Large Fish Collection Area

USEPA's Preliminary Site Perimeter

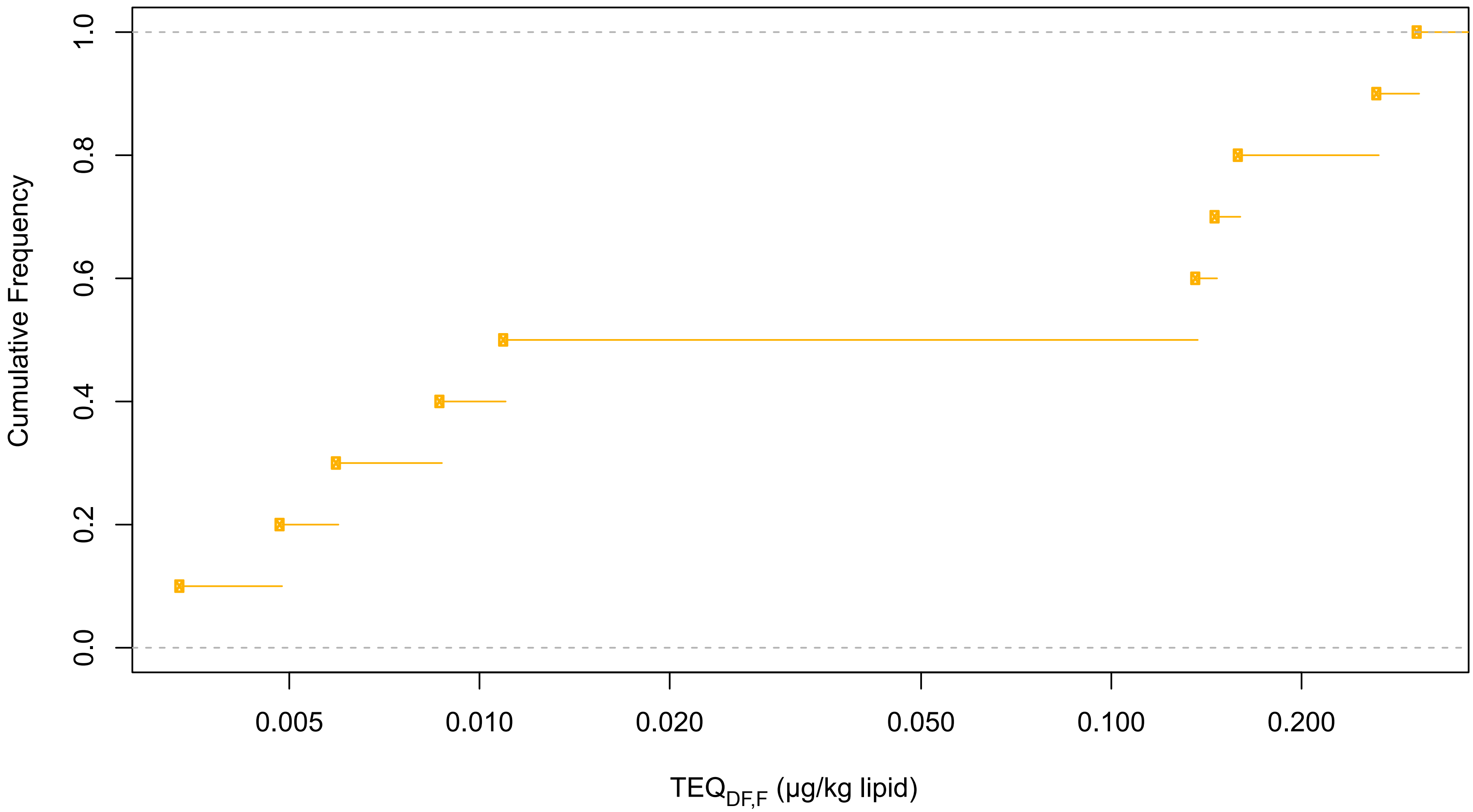
0

0.5

Scale in Miles

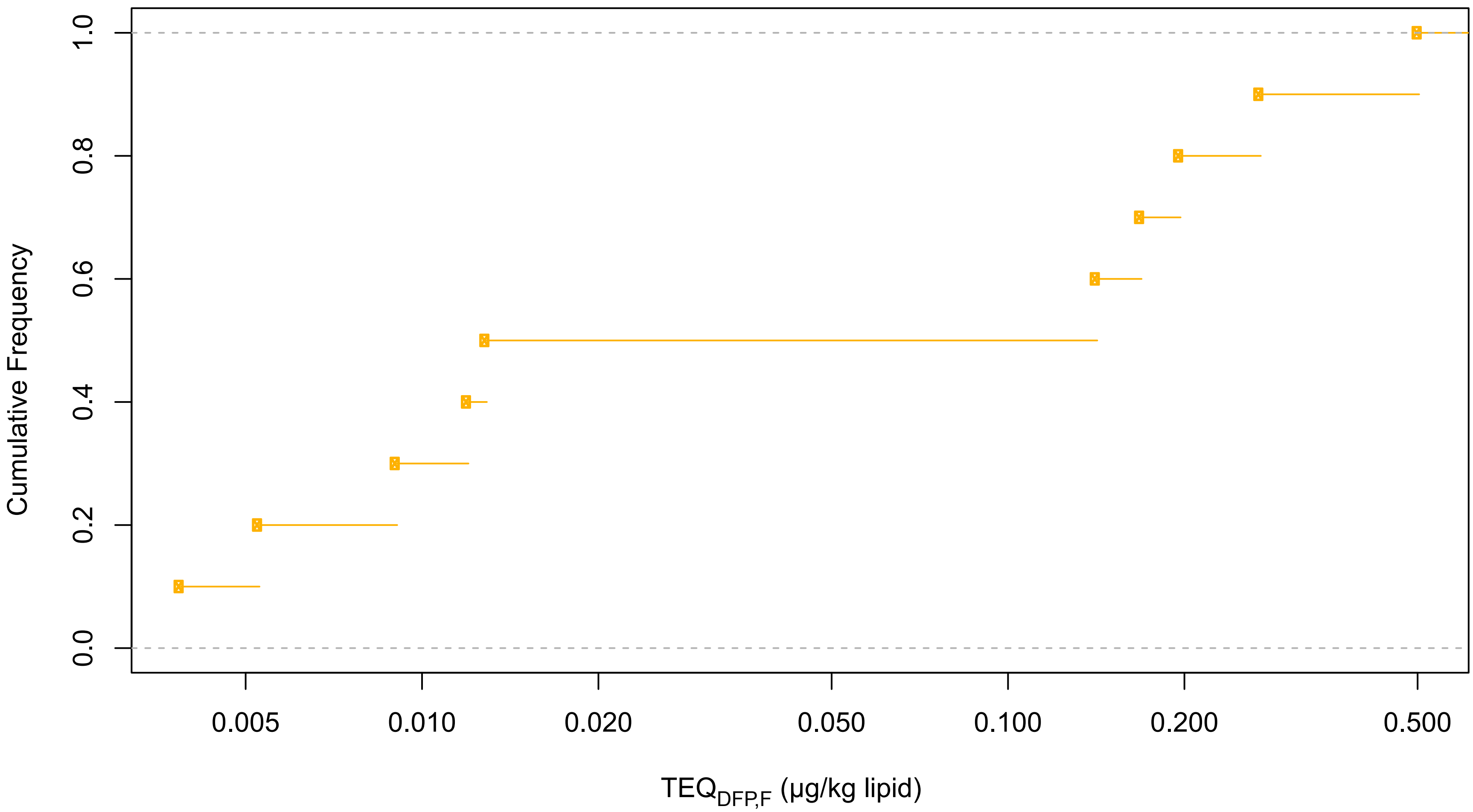
FEATURE SOURCES:
Aerial: ESRI USA Prime Imagery, 2008
Transportation Lines: ESRI World Transportation

Figure 4-3
Cedar Bayou Background Tissue Sampling Locations
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



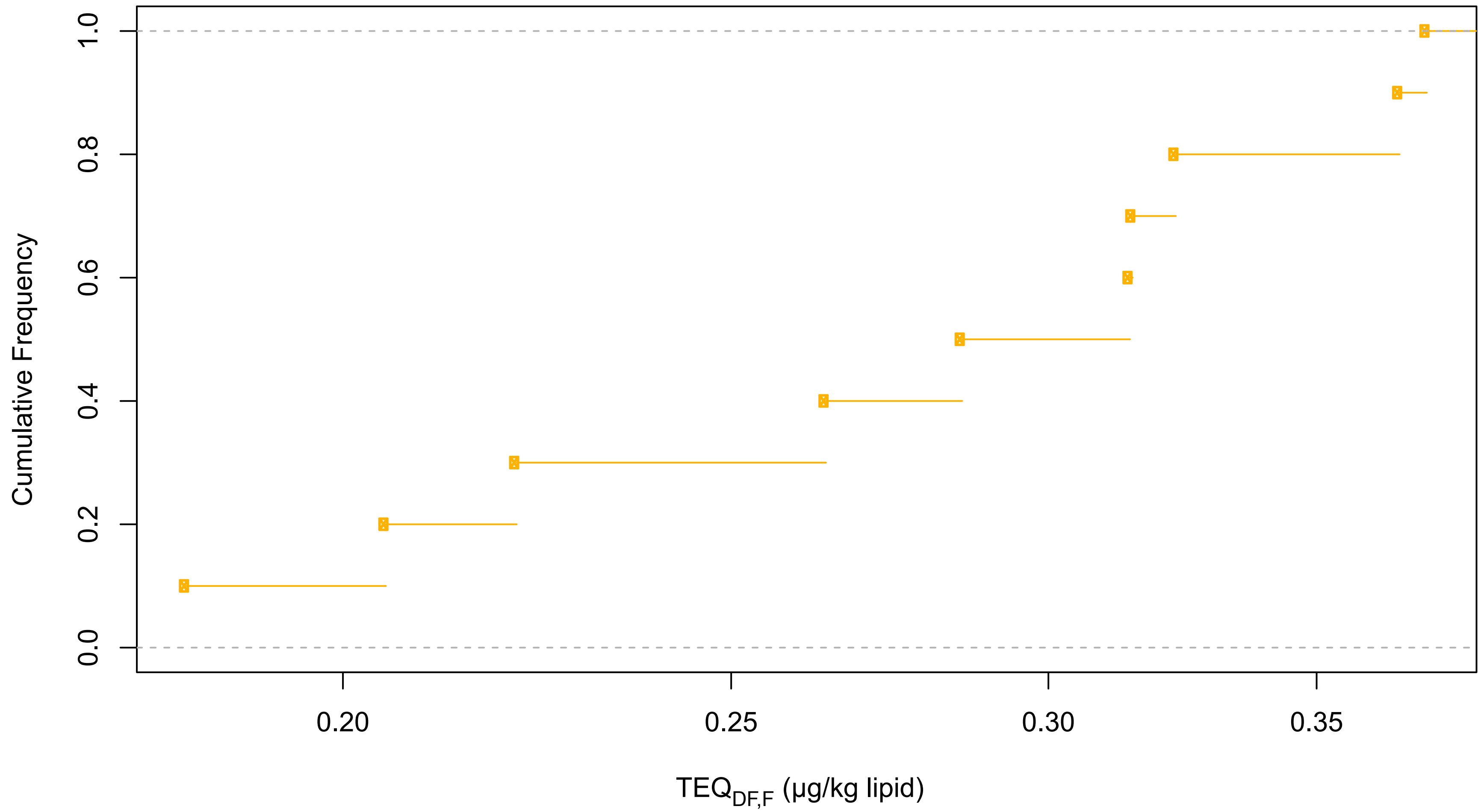
Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-4
Cumulative Frequency Distribution of Site Gulf Killifish Data ($TEQ_{DF,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



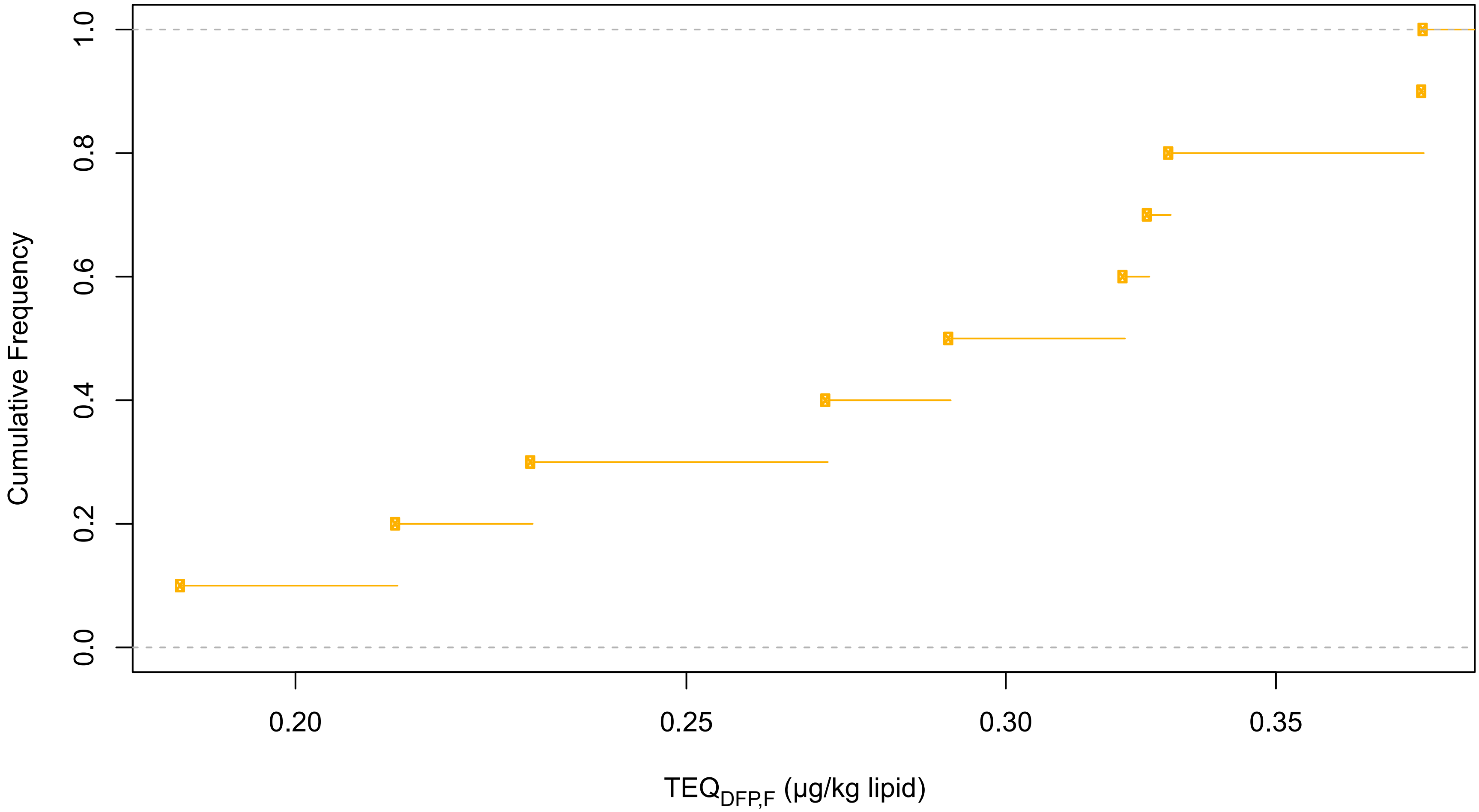
Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-5
Cumulative Frequency Distribution of Site Gulf Killifish Data (TEQ$_{DFP,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-6
Cumulative Frequency Distribution of Site Hardhead Catfish Data (TEQ$_{DF,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 4-7
Cumulative Frequency Distribution of Site Hardhead Catfish Data (TEQ$_{DFP,F}$)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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Hardhead Catfish Sample Locations

Fish-Crab Sample Area

Preliminary Site Perimeter

0 1,000

Feet

FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009
DOQQs –Texas Strategic Mapping
Program (StratMap), TNRIS

Figure 4-8
Hardhead Catfish Sample Locations
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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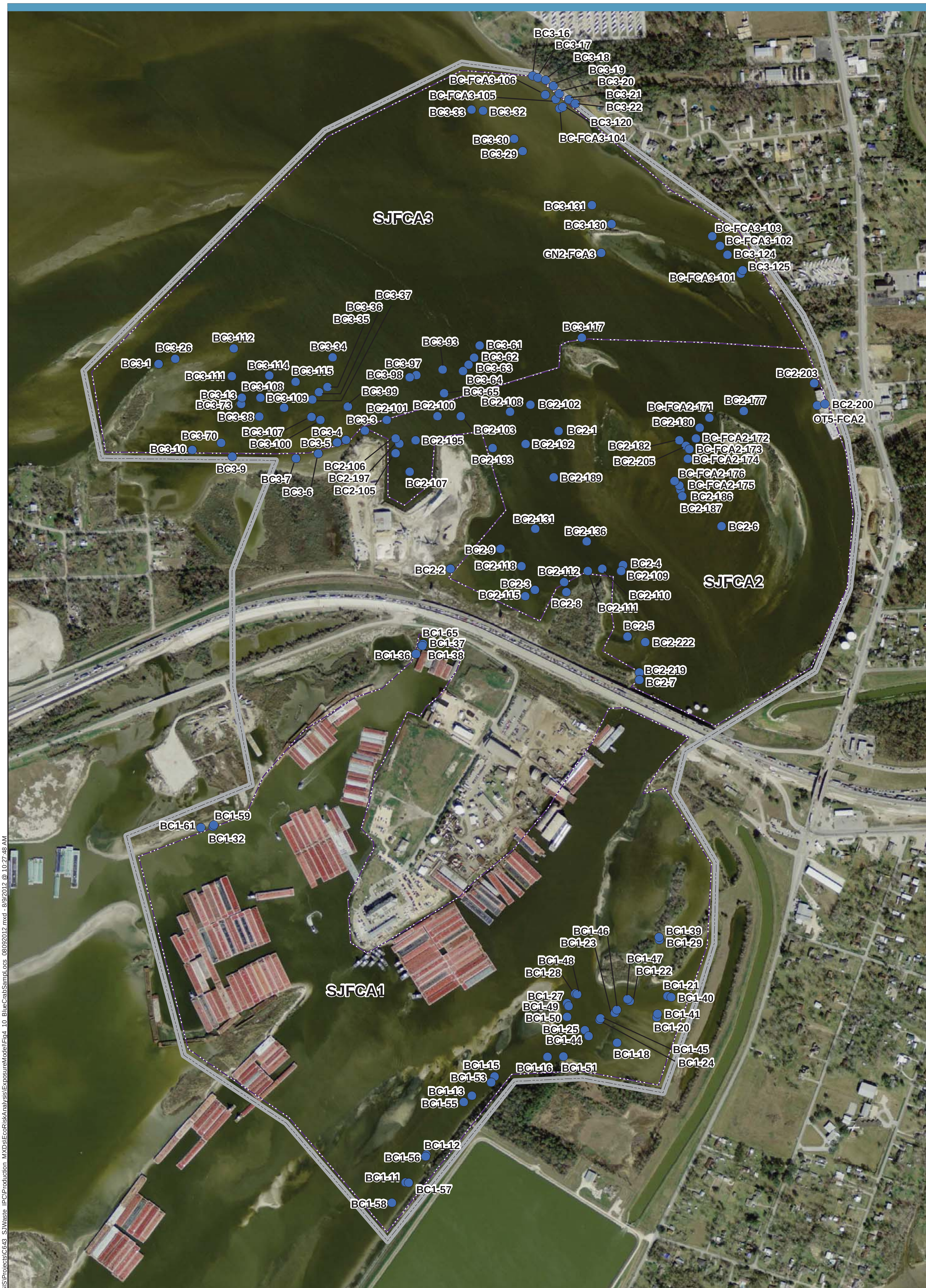
Hardhead Catfish Sample Locations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: ESRI USA Prime Imagery, 2009

Figure 4-9
Hardhead Catfish Sample Locations
Within Cedar Bayou
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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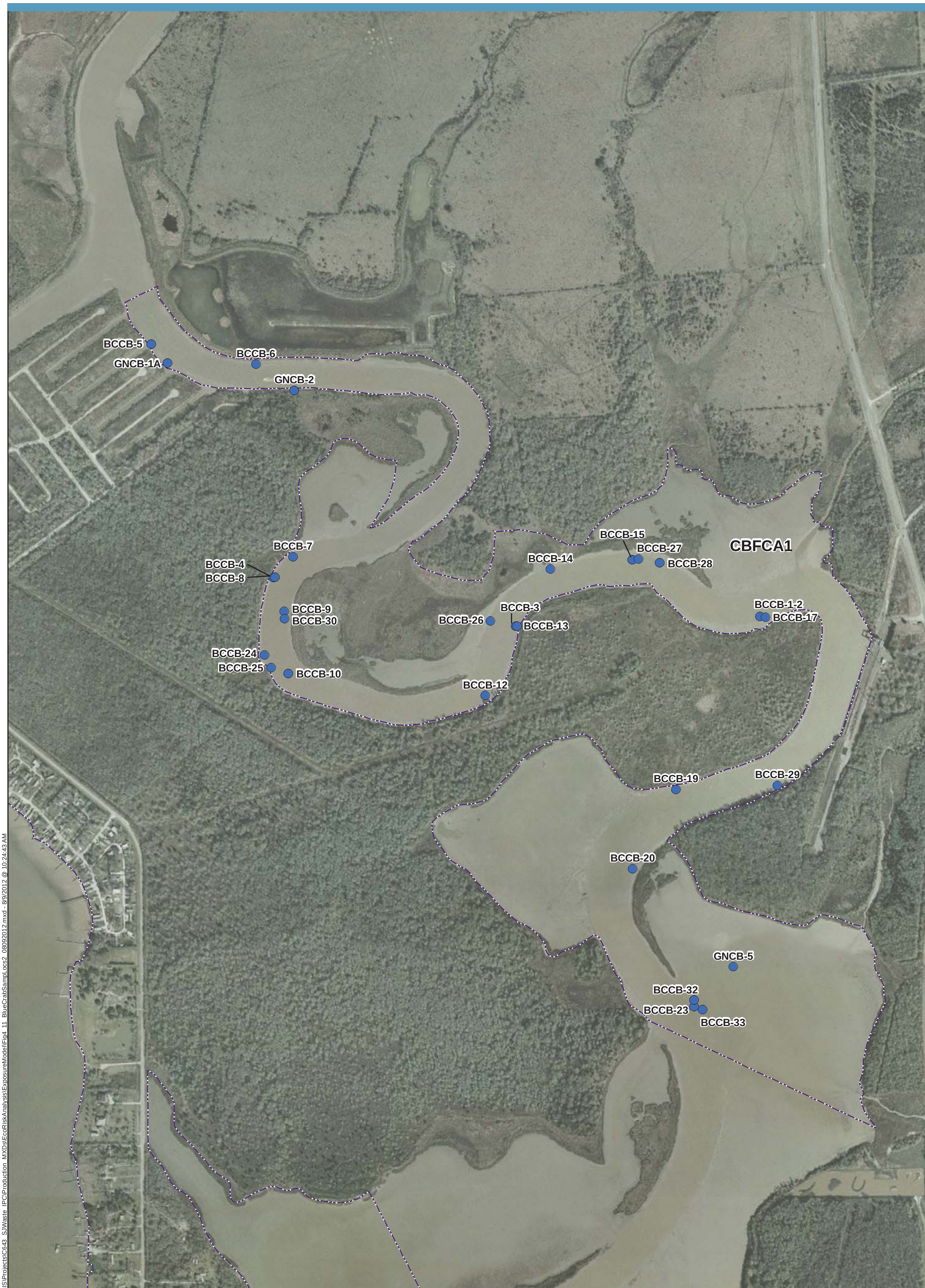
Blue Crab Sample Stations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: 0.5-meter 2008/2009
DOQQs - Texas Strategic Mapping
Program (StratMap), TNRIS

Figure 4-10
Locations of Blue Crab Collections
Within USEPA's Preliminary Site Perimeter
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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Blue Crab Sample Stations
Fish-Crab Sample Area
Preliminary Site Perimeter

0 1,000
Feet

FEATURE SOURCES:
Aerial Imagery: ESRI USA Prime Imagery, 2009

Figure 4-11
Locations of Blue Crab Collections
Within Cedar Bayou
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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0 1,000

Feet

Surface Soil Sample Location (0 - 6 inches)

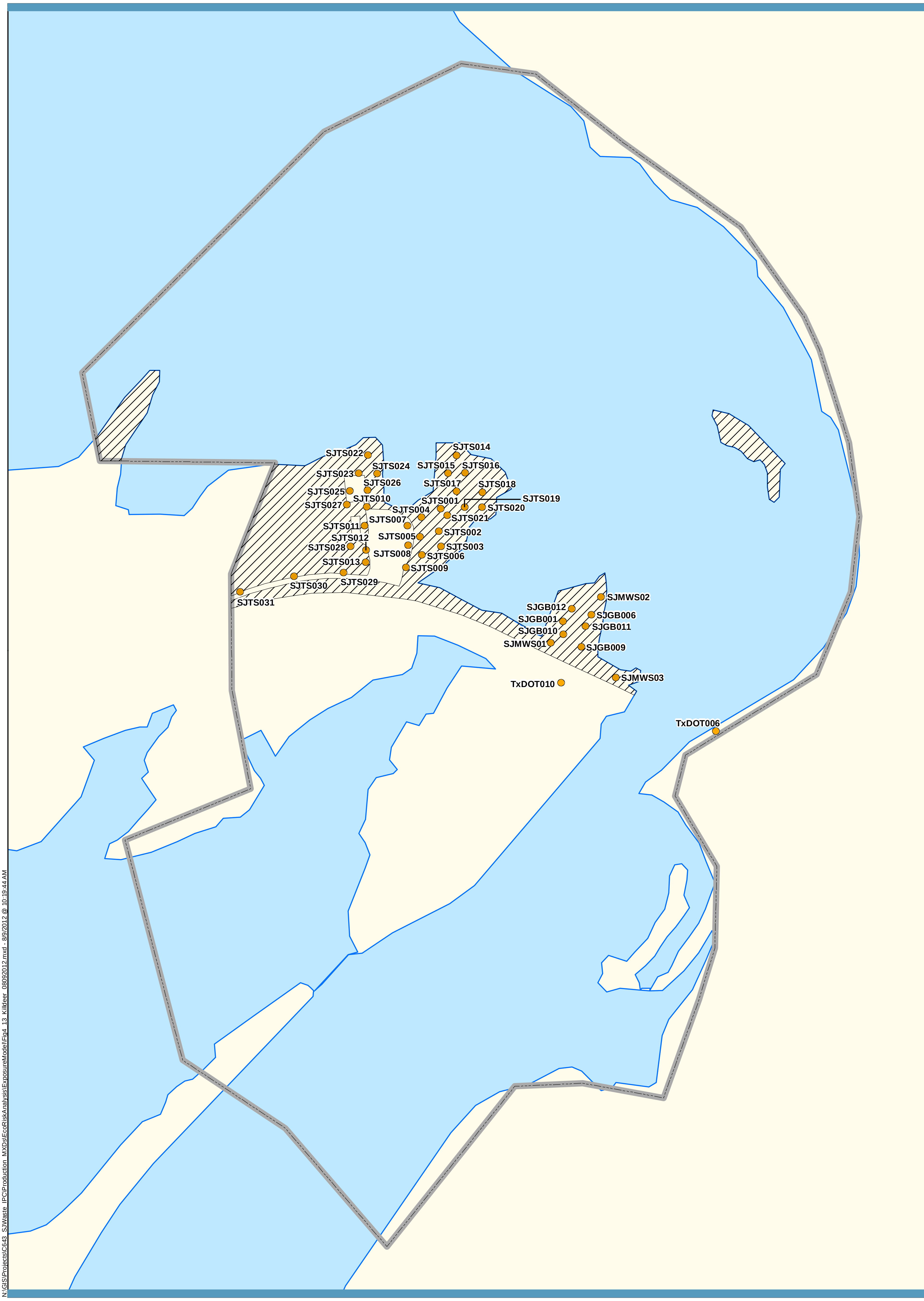
- 0.0005 - 3.2
- 3.2 - 6.5
- 6.5 - 9.7
- 9.7 - 12.9
- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 4-12

Concentrations of Mercury (mg/kg) in Surface Soils within the Preliminary Site Perimeter

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



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0 1,000

Feet

Soil Sample Locations North of I-10 (0 to 6 inches)

Killdeer Estimated Exposure Area

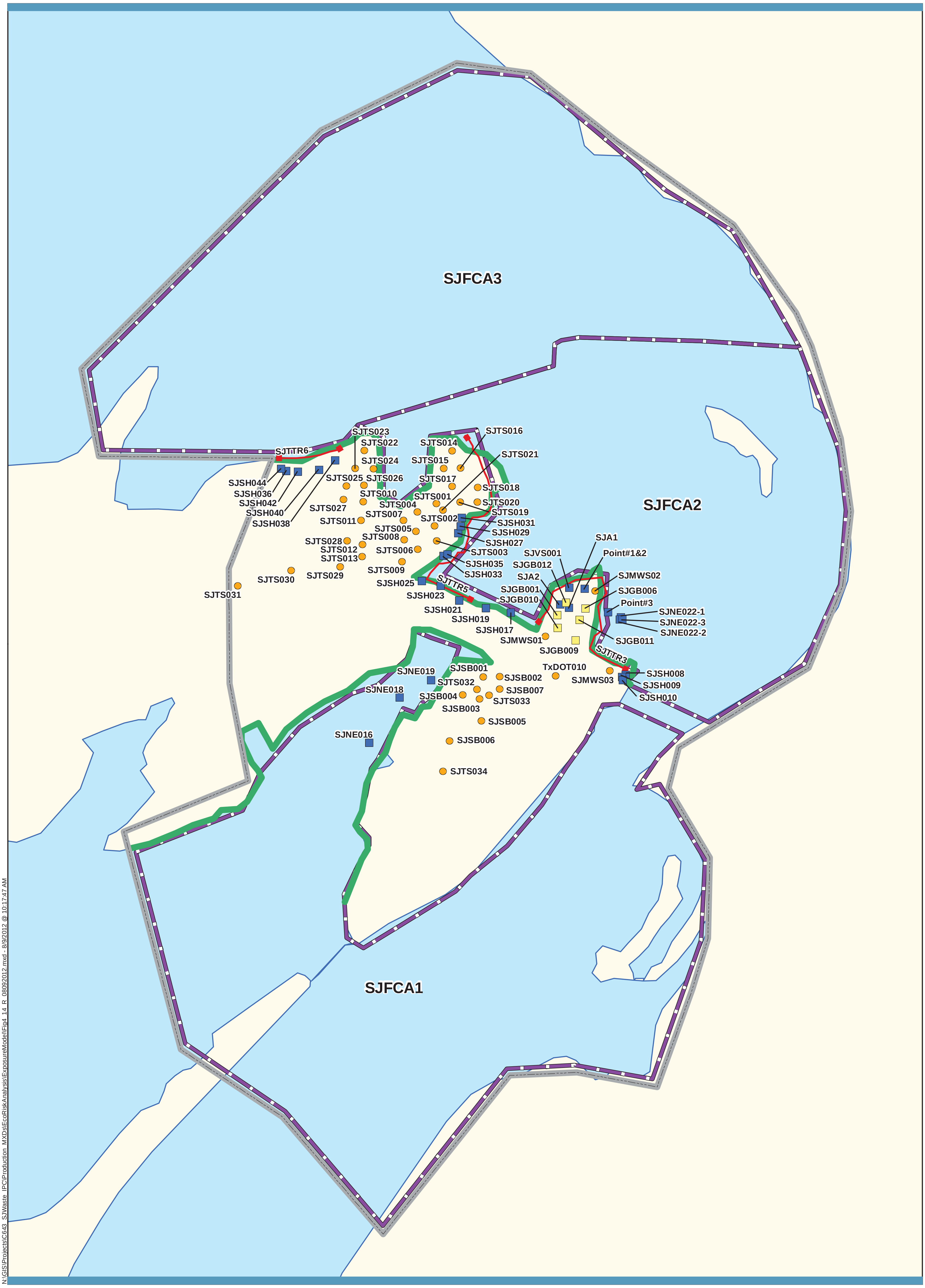
Preliminary Site Perimeter

Figure 4-13

Exposure Areas and Samples Used for Estimating Exposures to Killdeer

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



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0 1,000
Feet

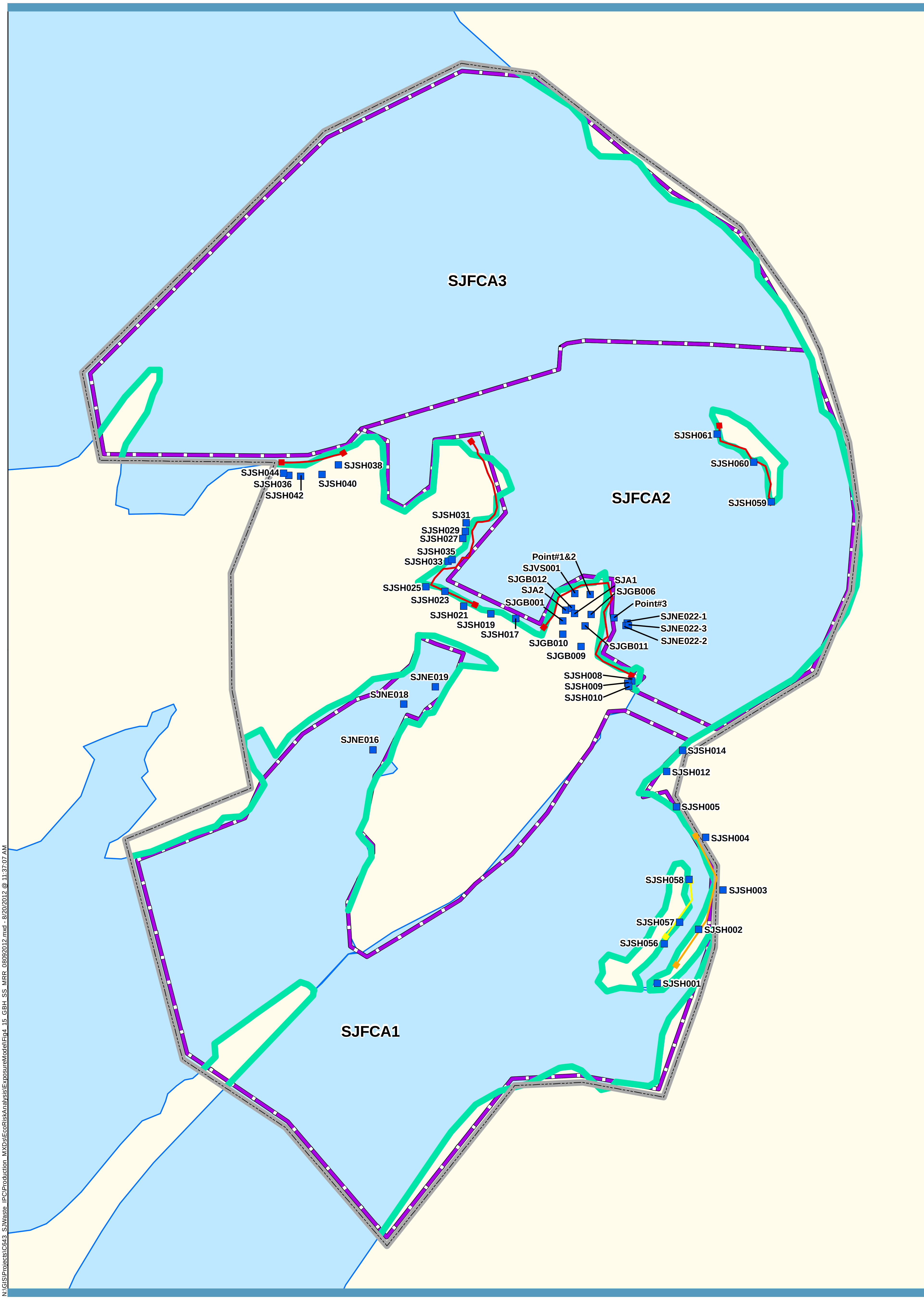
Shoreline Surface Sediment Sample Locations (0-6 inches)

- Used as Sediment to Evaluate Exposures
- Used as Both Sediment and Soil to Evaluate Exposures

Surface Soil Sample Locations (0 to 6 inches)

- Clams, Seines (Small Fish), and Infauna
- Raccoon Estimated Exposure Area
- Catfish / Blue Crab Sample Area
- Preliminary Site Perimeter

Figure 4-14
Exposure Areas and Samples Used
for Estimating Exposures to Raccoons
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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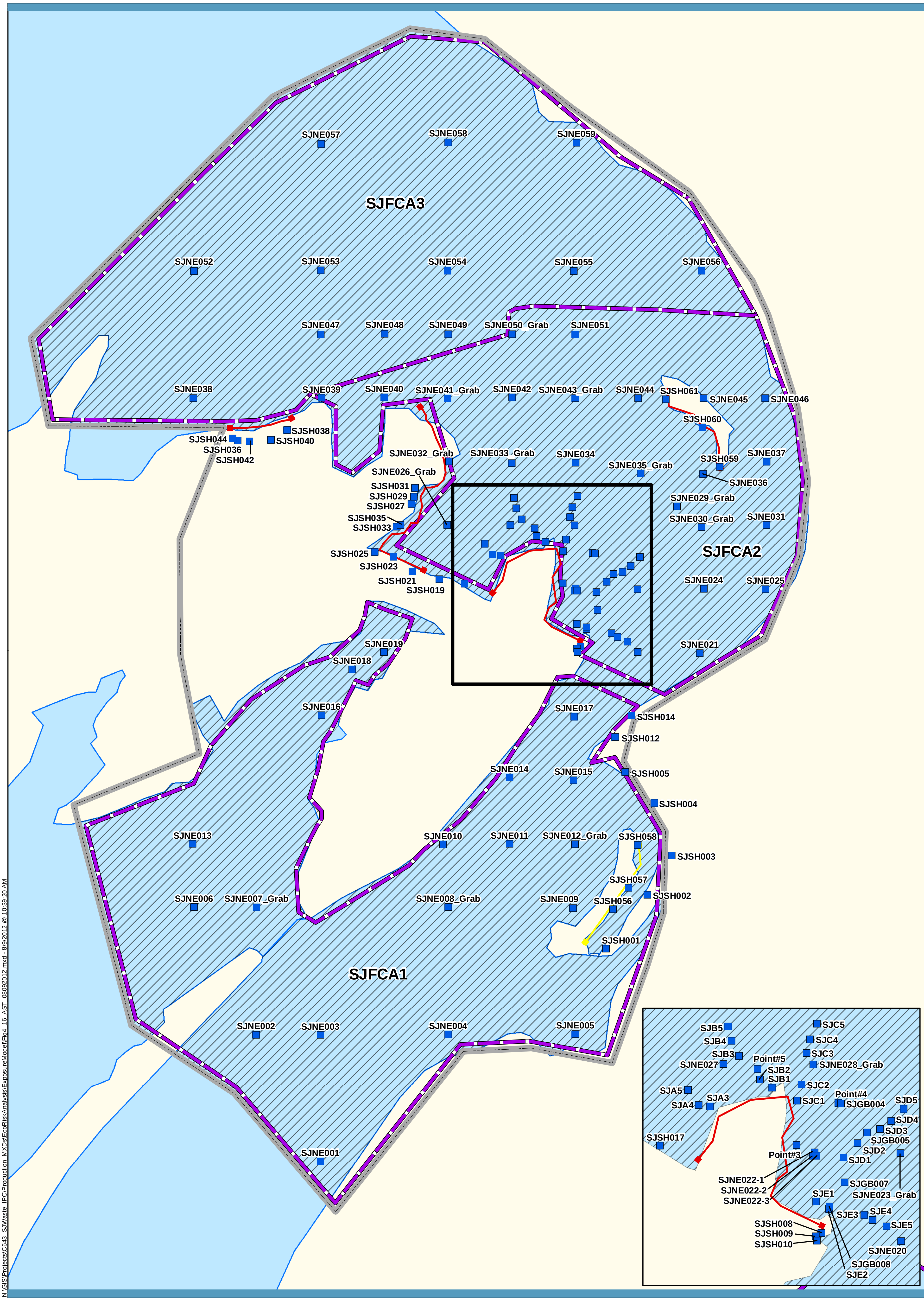
Great Blue Heron, Spotted Sandpiper and Marsh Rice Rat Estimated Exposure Areas
Catfish / Blue Crab Sample Area
Sampling Transects
Clams
Clams, Seines (Small Fish), and Infauna
Seines and Infauna
Shoreline Surface Sediment Sample Locations (0-6 inches)
Preliminary Site Perimeter

0 1,000
Feet

Figure 4-15
Exposure Areas and Samples Used for Estimating Exposures to Great Blue Herons, Spotted Sandpipers, and Marsh Rice Rats
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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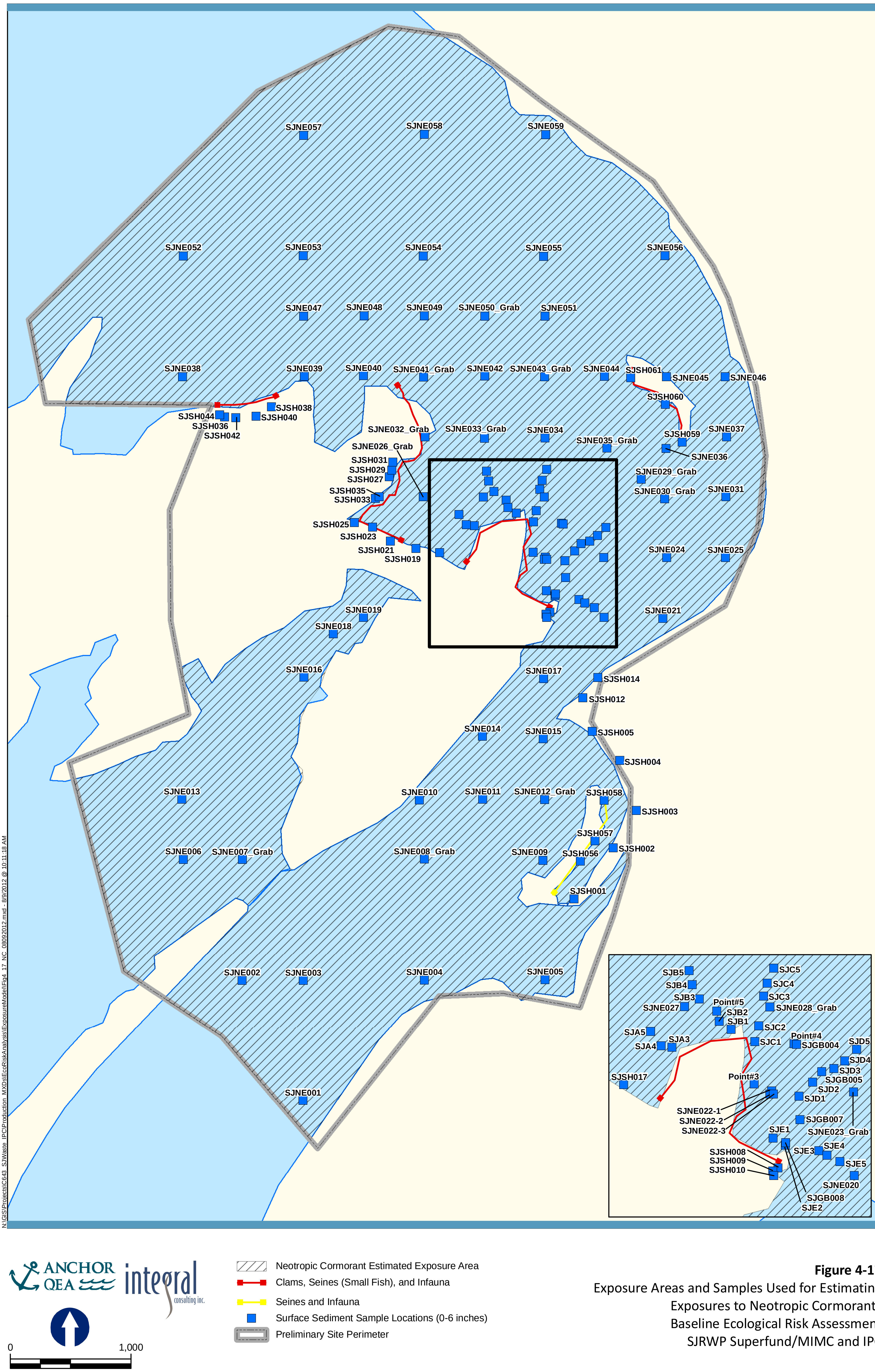
0 1,000

Feet

- Surface Sediment Sample Locations (0-6 inches)
- Clams, Seines (Small Fish), and Infauna
- Seines and Infauna
- Preliminary Site Perimeter
- Catfish / Blue Crab Sample Area
- Alligator Snapping Turtle Estimated Exposure Area

Figure 4-16
Exposure Areas and Samples Used for Estimating Exposures to Alligator Snapping Turtles
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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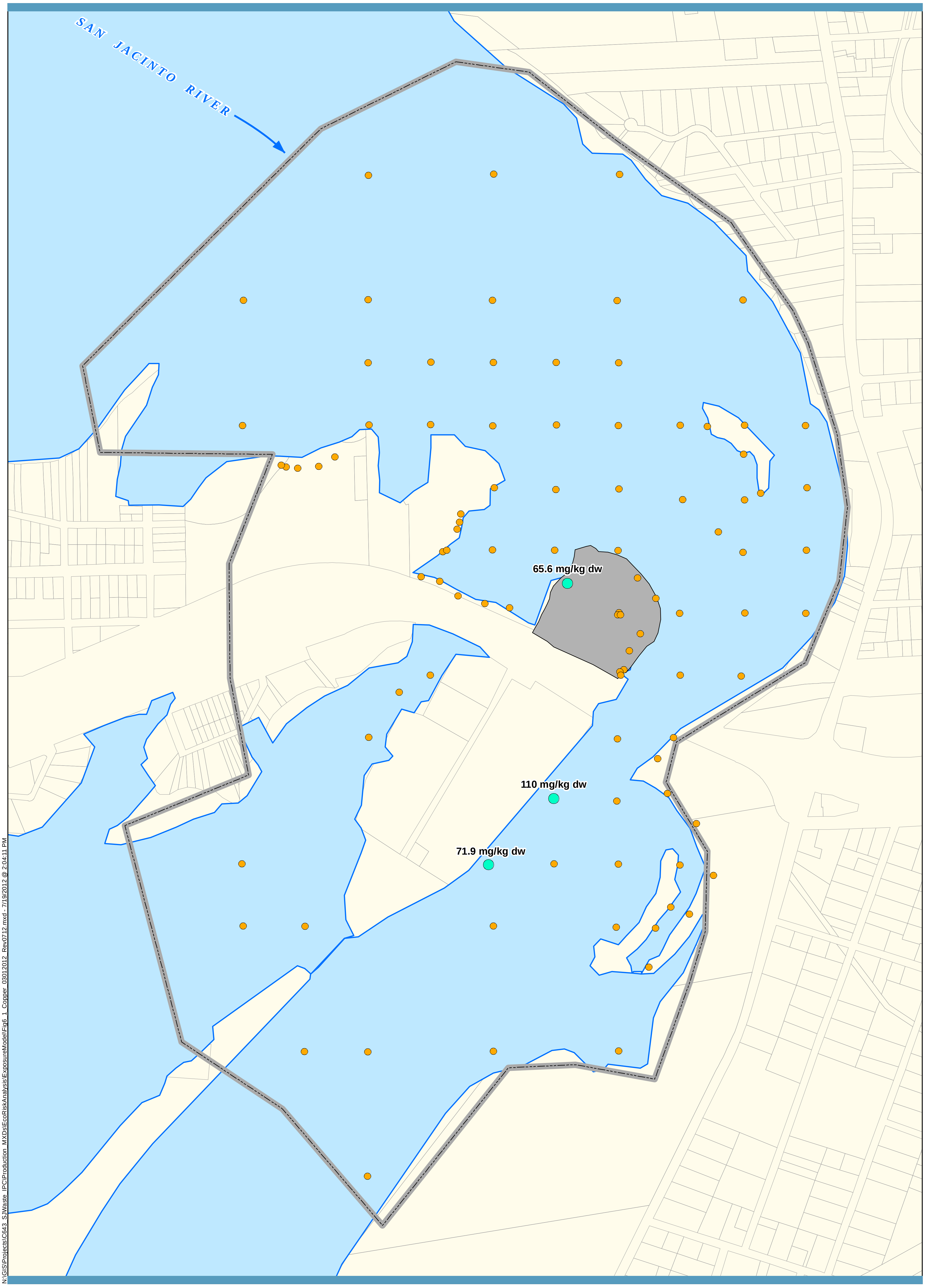
- Neotropic Cormorant Estimated Exposure Area
- Clams, Seines (Small Fish), and Infauna
- Seines and Infauna
- Surface Sediment Sample Locations (0-6 inches)
- Preliminary Site Perimeter

Figure 4-17

Exposure Areas and Samples Used for Estimating Exposures to Neotropic Cormorants

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



ANCHOR QEA integral consulting inc.

0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed ERL or ERM

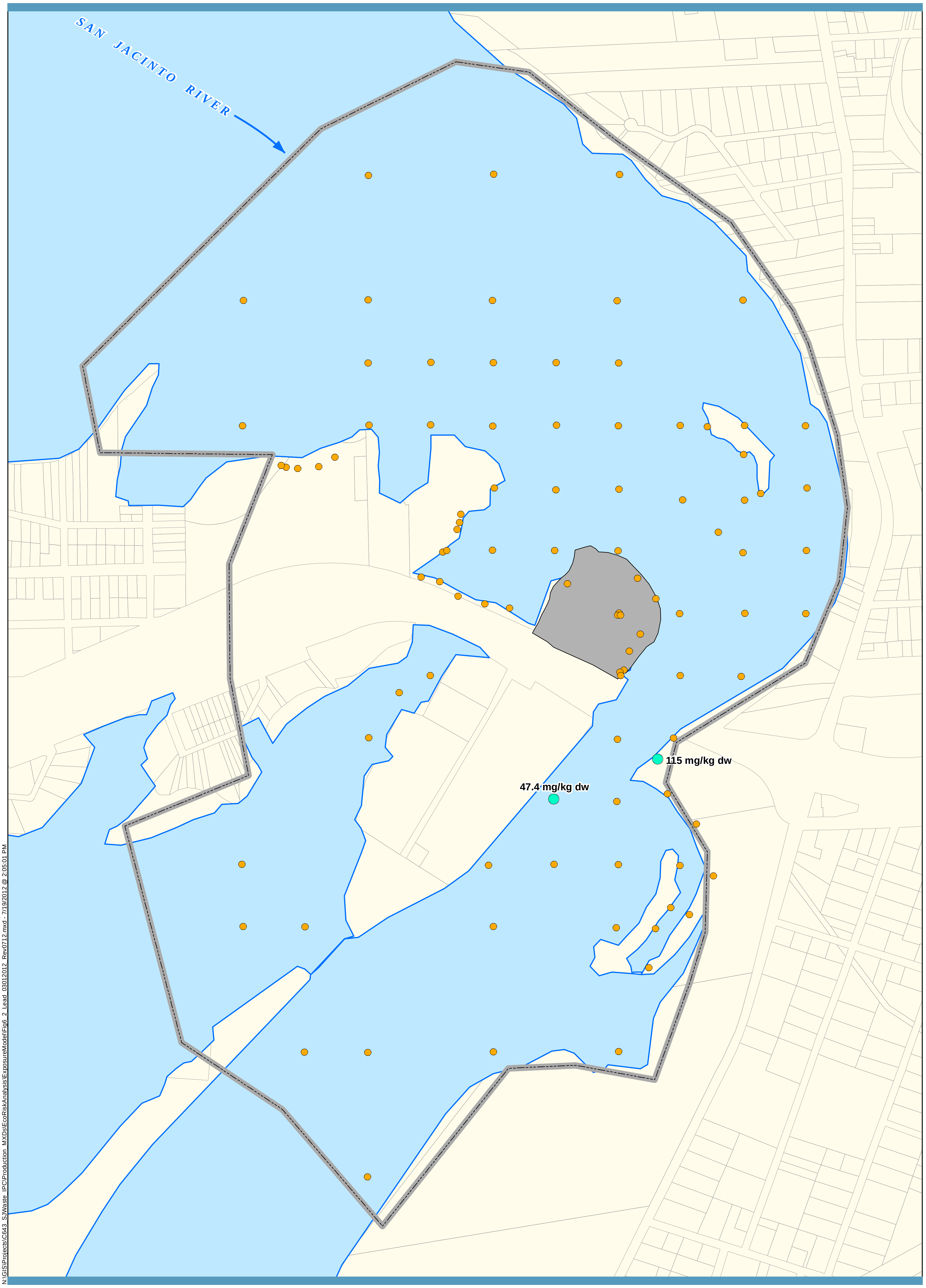
Exceeds ERL (Does Not Exceed ERM)

Copper Effects Range Low (ERL) = 34 mg/kg
Copper Effects Range Medium (ERM) = 270 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-1
Concentrations of Copper in Sediment Relative to the SQG for Copper (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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ANCHOR QEA
integral consulting inc.

0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed ERL or ERM

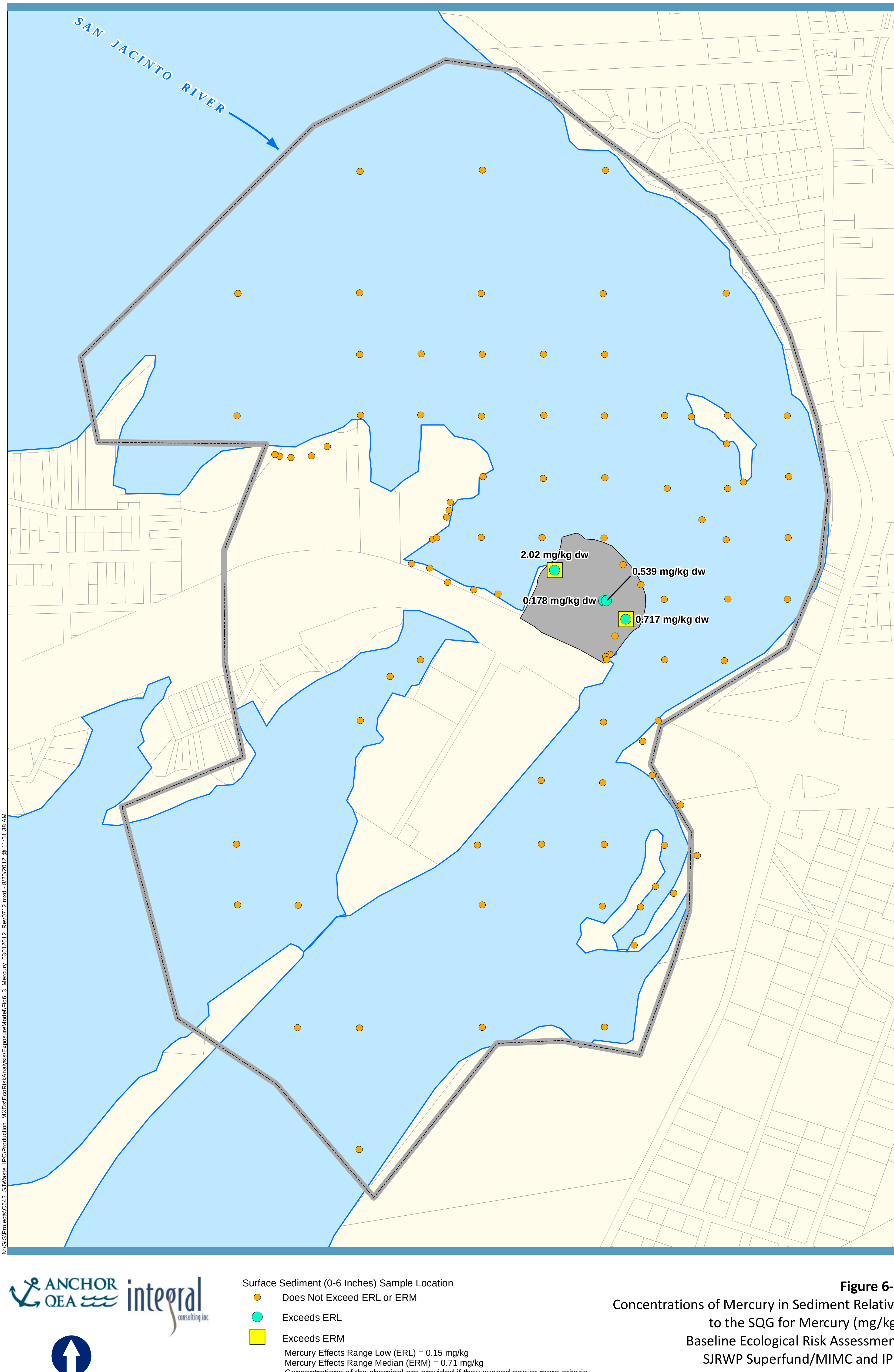
Exceeds ERL (Does Not Exceed ERM)

Lead Effects Range Low (ERL) = 47 mg/kg
Lead Effects Range Medium (ERM) = 218 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-2
Concentrations of Lead in Sediment Relative to the SQG for Lead (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA
integral consulting inc.

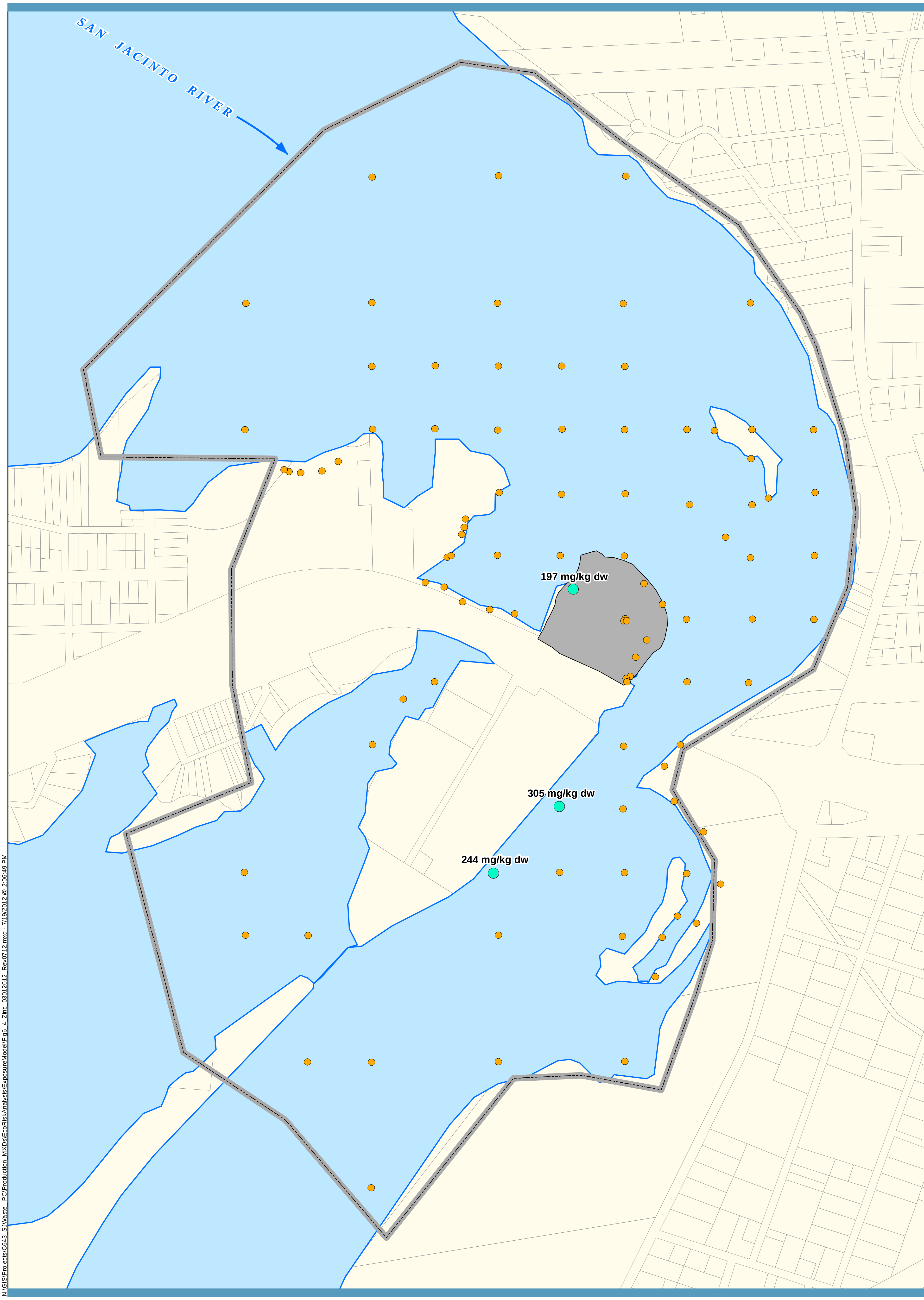
0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location
- Does Not Exceed ERL or ERM
- Exceeds ERL
- Exceeds ERM

Mercury Effects Range Low (ERL) = 0.15 mg/kg
Mercury Effects Range Median (ERM) = 0.71 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

USEPA's Preliminary Site Perimeter
Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-3
Concentrations of Mercury in Sediment Relative to the SQG for Mercury (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed ERL or ERM

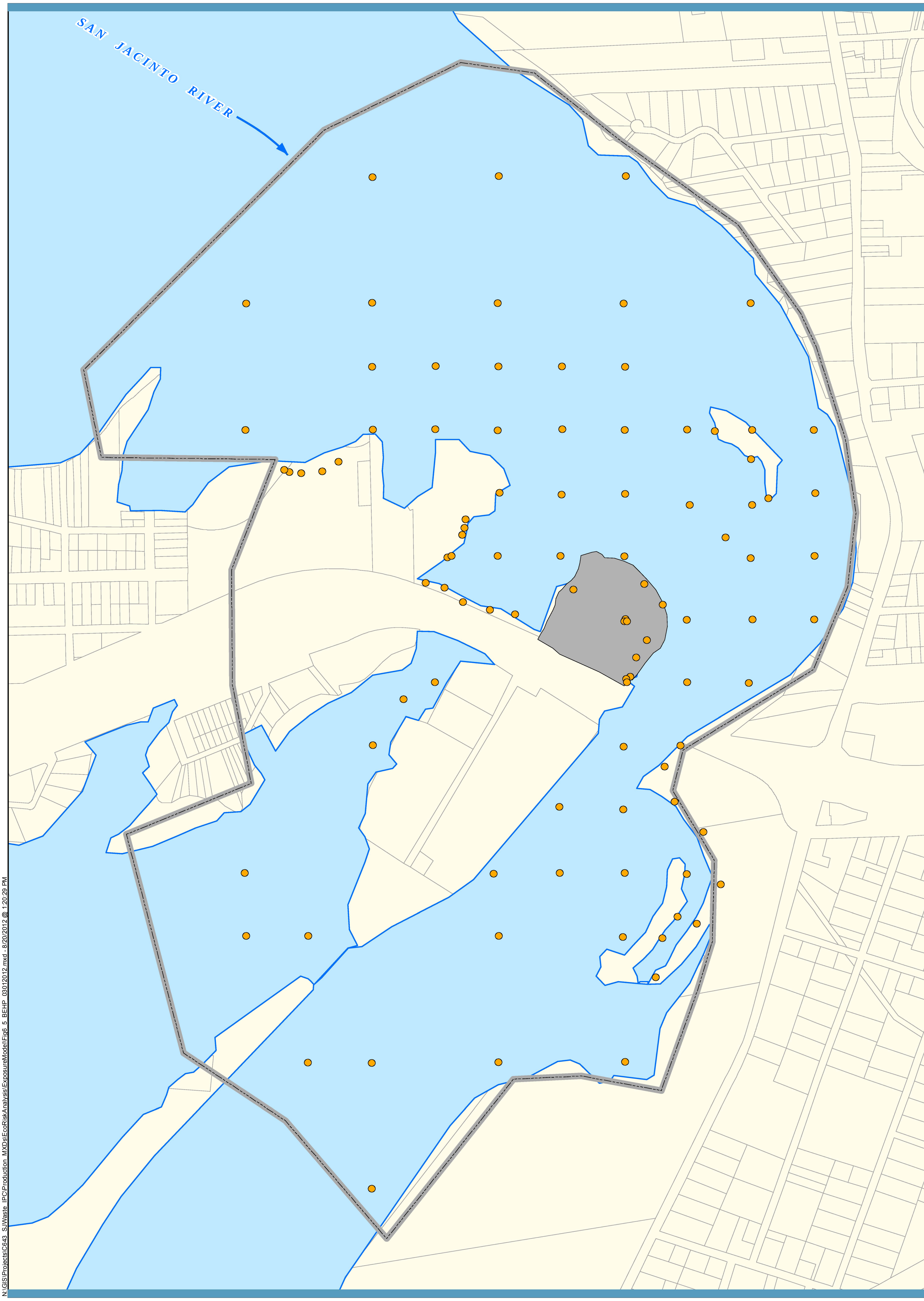
Exceeds ERL (Does Not Exceed ERM)

Zinc Effects Range Low (ERL) = 150 mg/kg
Zinc Effects Range Medium (ERM) = 410 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-4
Concentrations of Zinc in Sediment Relative to the SQG for Zinc (mg/kg)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

Bis(2-ethylhexyl)phthalate Surface Water Toxicity Reference Value (TRV) = 100 µg/L

USEPA's Preliminary Site Perimeter

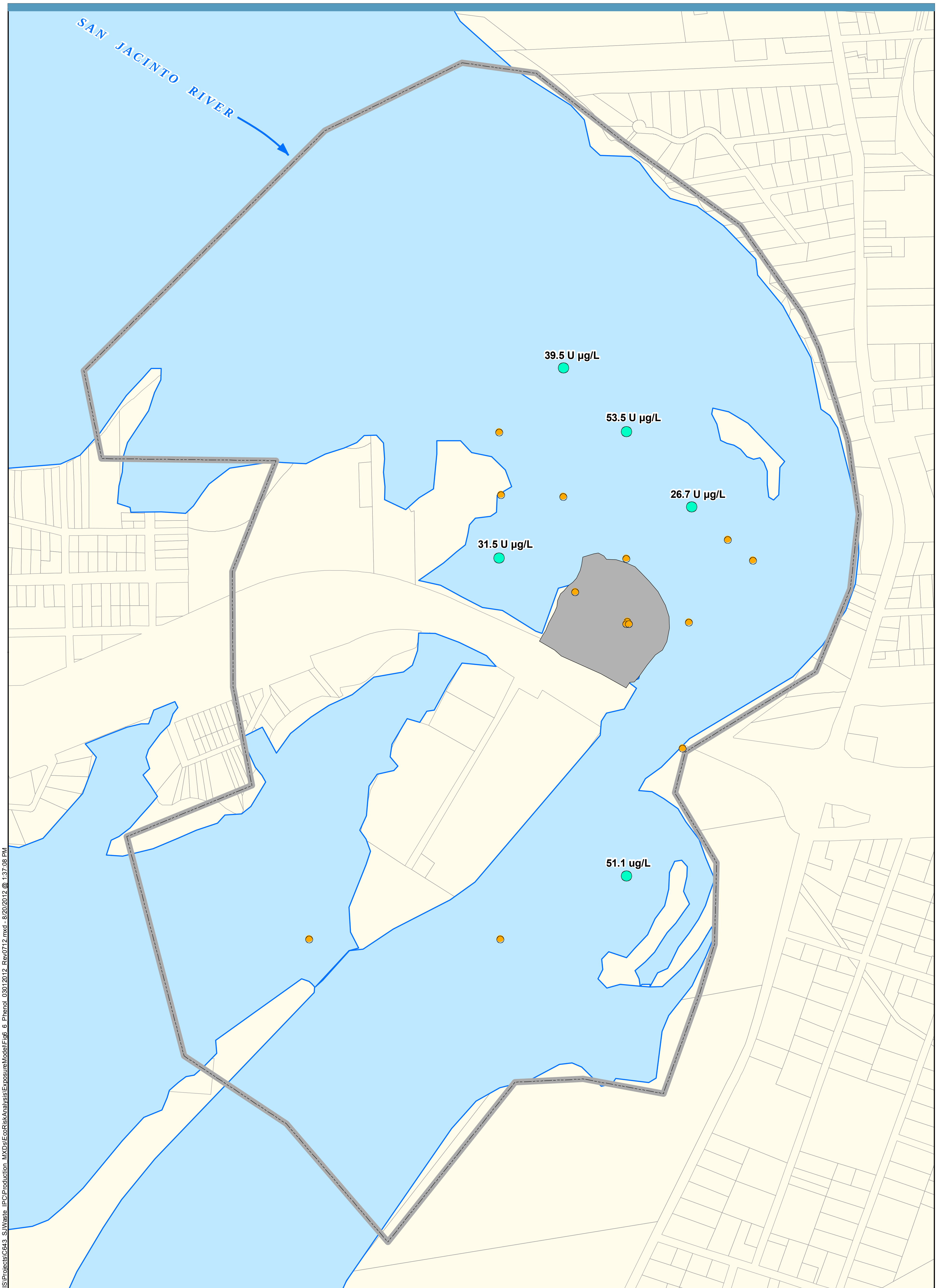
Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-5

Estimated Porewater Concentrations of Bis(2-ethylhexyl)phthalate (BEHP) Relative to the TRV for BEHP (µg/L)

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



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ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

Exceeds TRV

Phenol Toxicity Reference Value (TRV) = 26 µg/L
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-6
Estimated Porewater Concentrations of Phenol Relative to the TRV for Phenol (µg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

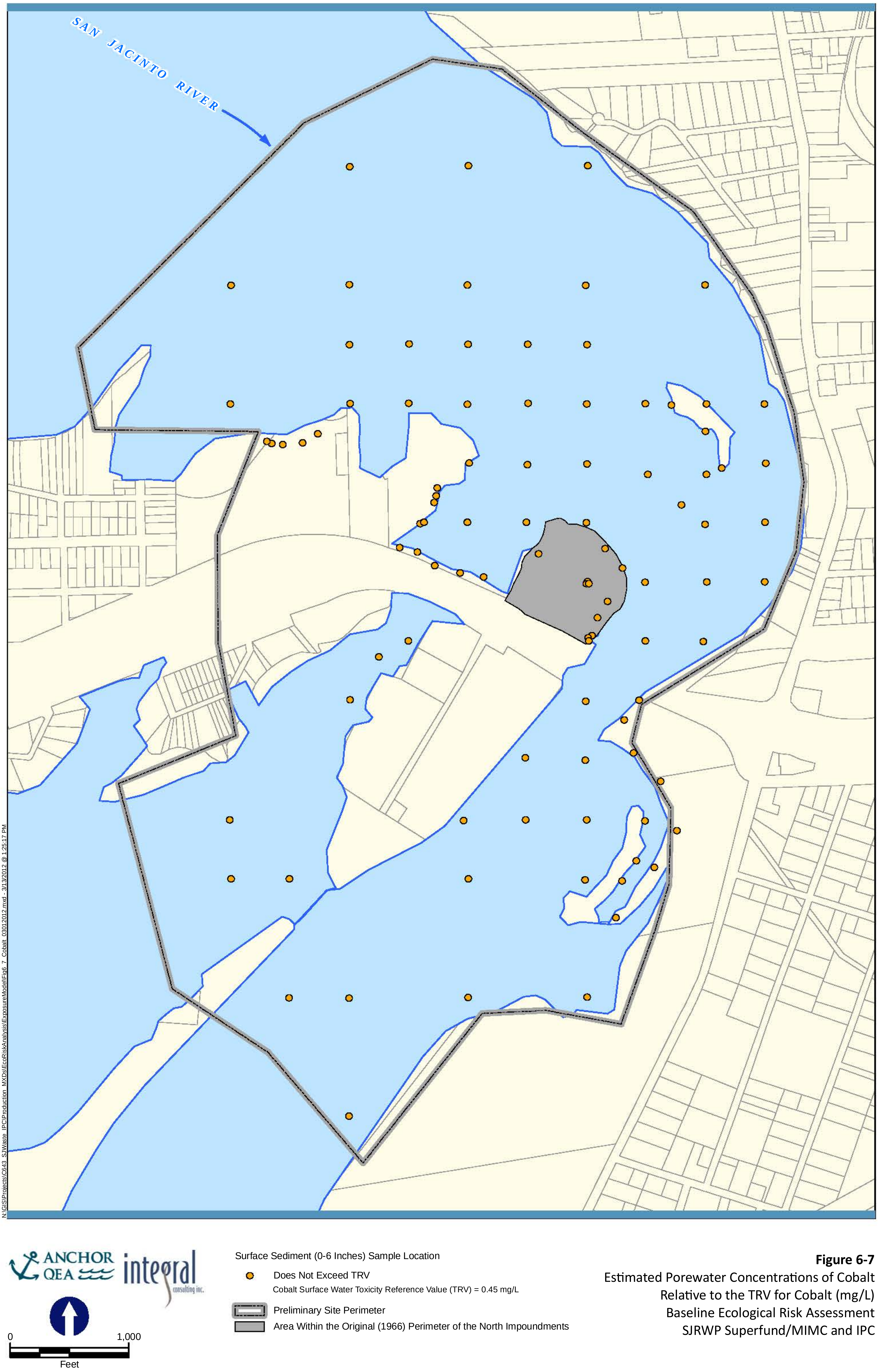
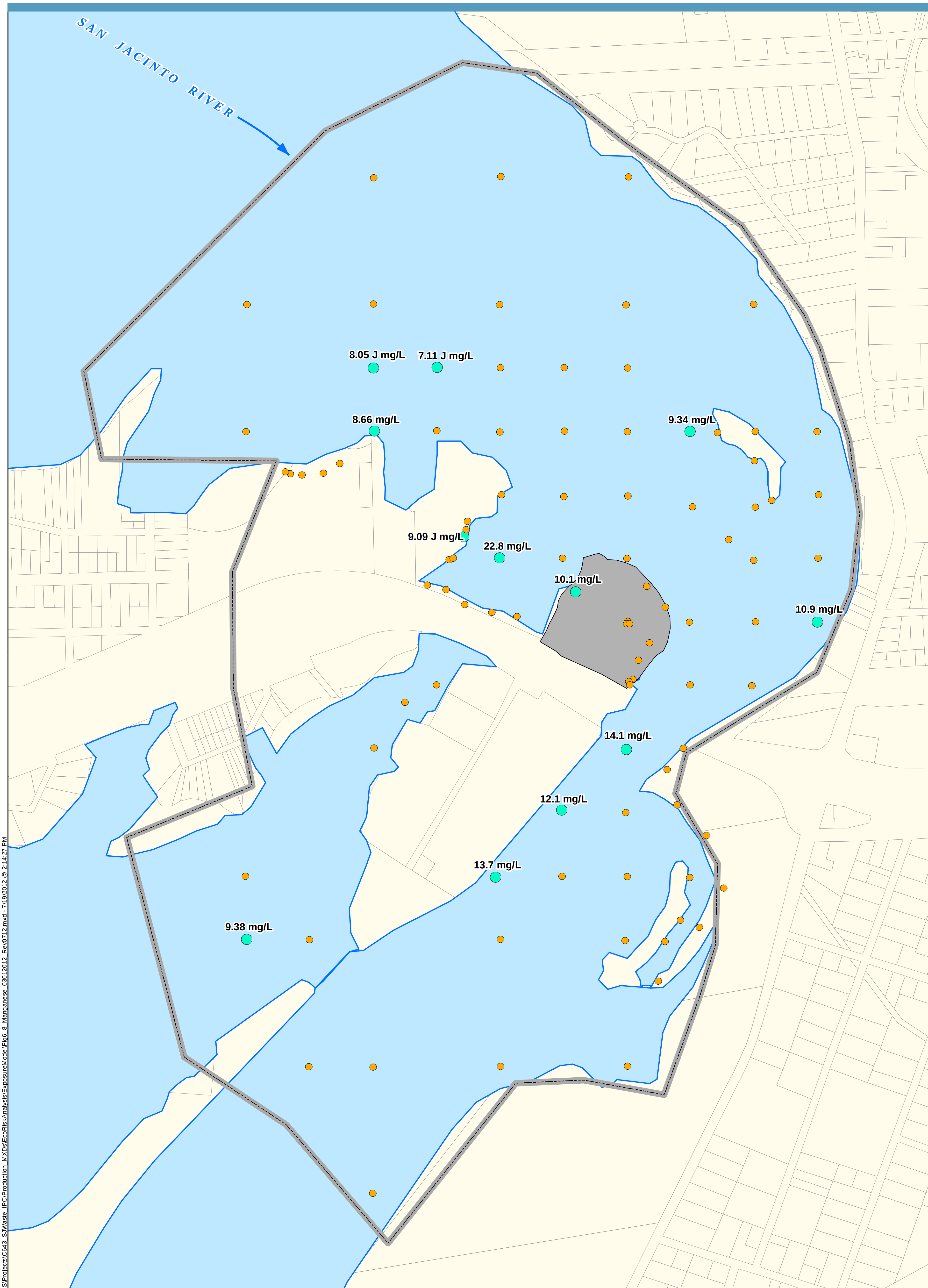


Figure 6-7
Estimated Porewater Concentrations of Cobalt Relative to the TRV for Cobalt (mg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

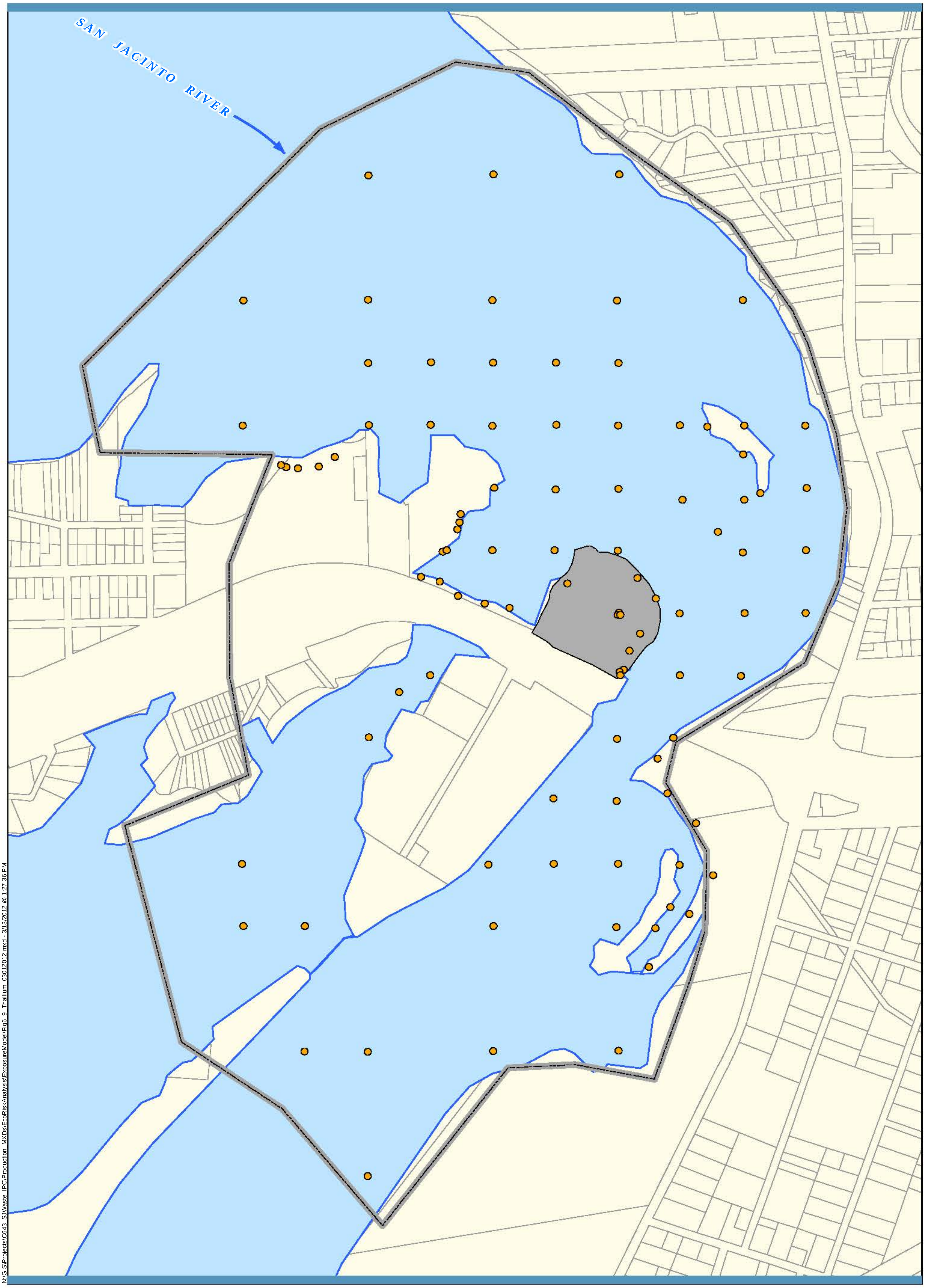
Exceeds TRV

Manganese Surface Water Toxicity Reference Value (TRV) = 7 mg/L
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-8
Estimated Porewater Concentrations of Manganese
Relative to the TRV for Manganese (mg/L)
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed TRV

Thallium Surface Water Toxicity Reference Value (TRV) = 0.213 mg/L

Preliminary Site Perimeter

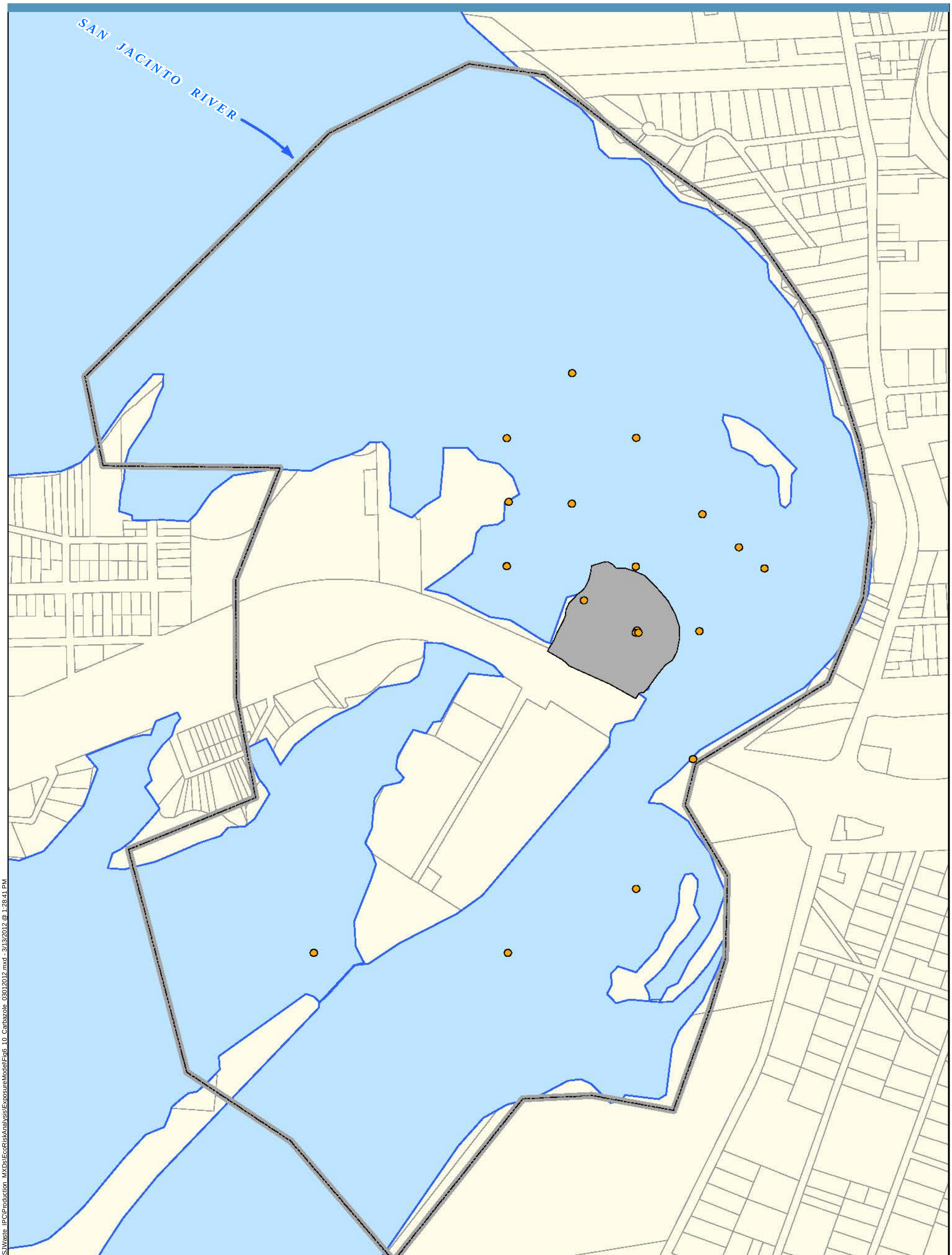
Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-9

Estimated Porewater Concentrations of Thallium Relative to the TRV for Thallium (mg/L)

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed Maximum Upstream Detection Limit

Carbazole Maximum Upstream Detection Limit = 23,300 ug/kg oc

Preliminary Site Perimeter

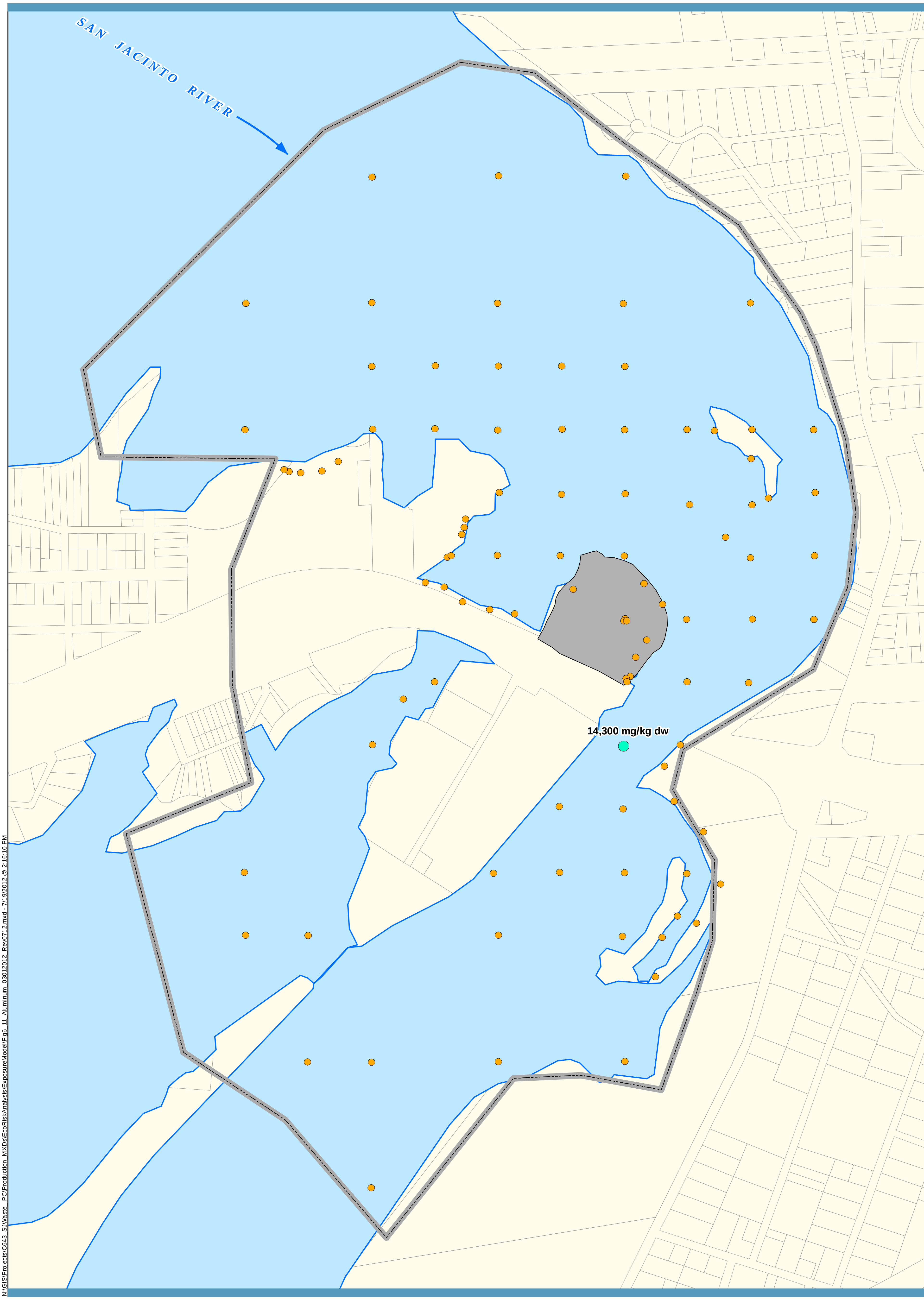
Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-10

Concentrations of Carbazole in Sediment Relative to the Upstream Maximum Detection Limit for Carbazole

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



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ANCHOR QEA
integral consulting inc.

0 1,000
Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed REV

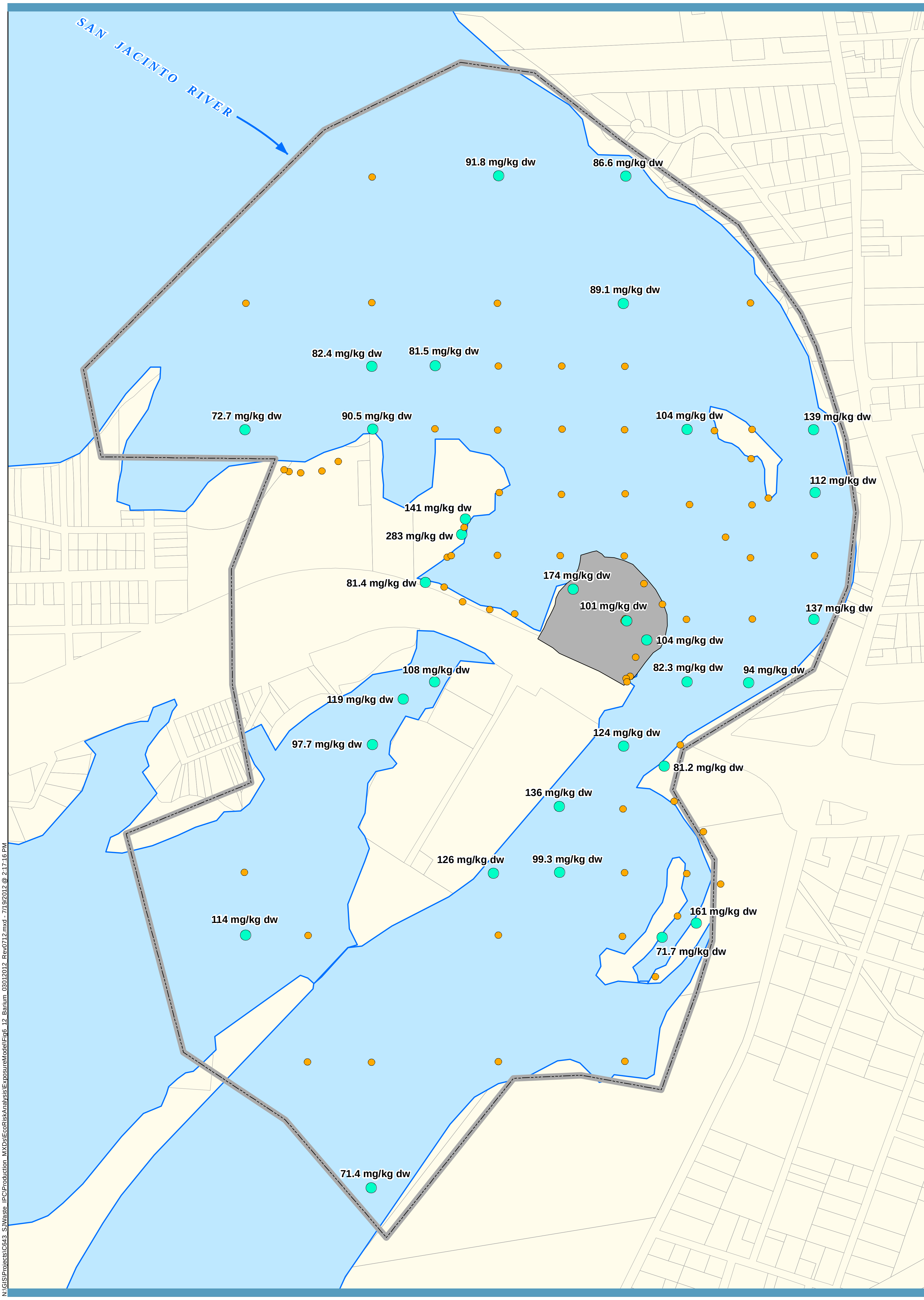
Exceeds REV

Aluminum Reference Envelope Value (REV) = 13,300 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-11
Concentrations of Aluminum in Sediment Relative to the Upstream Reference Envelope Value for Aluminum
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



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ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed REV

Exceeds REV

Barium Reference Envelope Value (REV) = 69.8 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

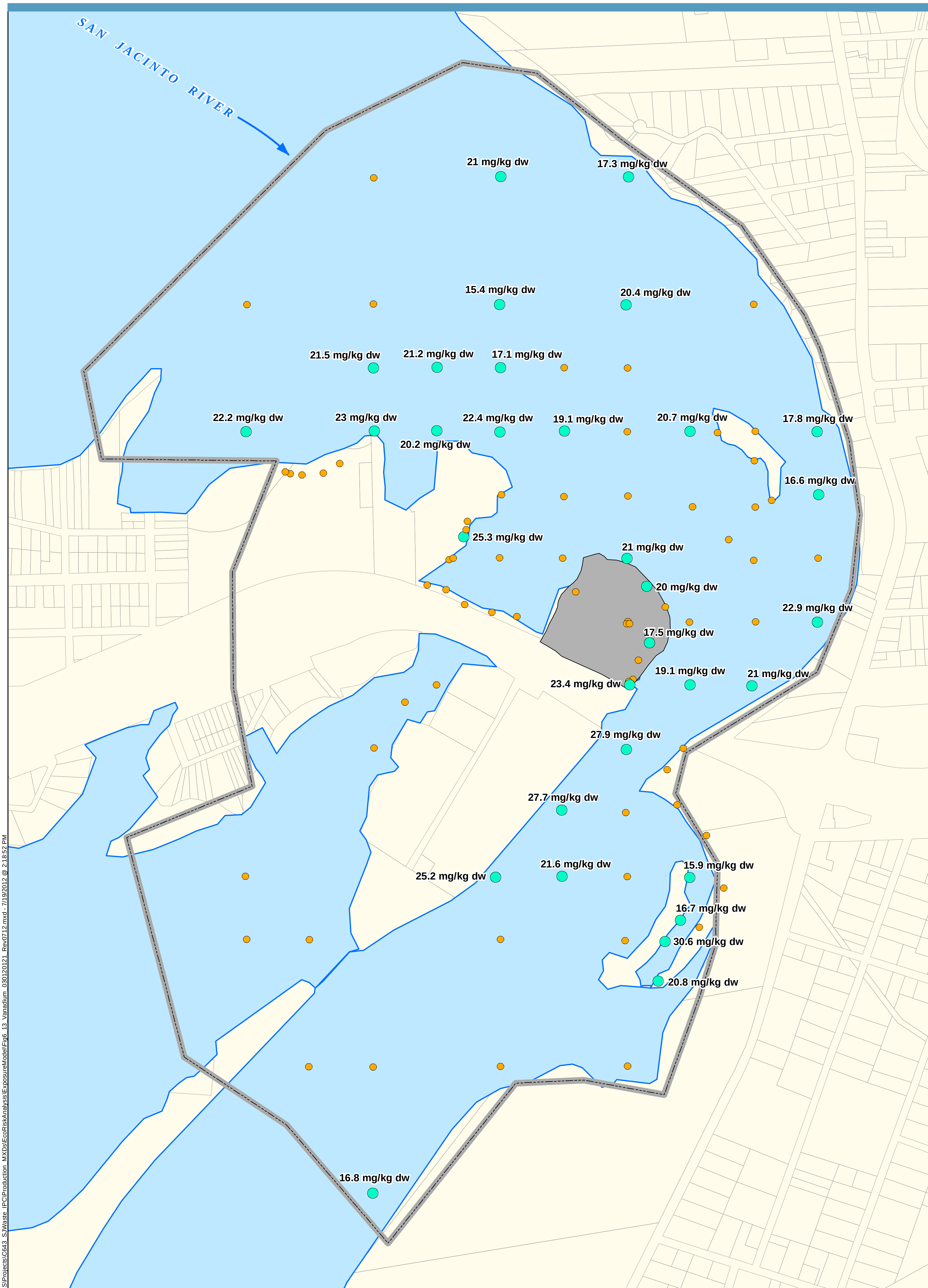
Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-12

Concentrations of Barium in Sediment Relative to the Upstream Reference Envelope Value for Barium

Baseline Ecological Risk Assessment

SJRWP Superfund/MIMC and IPC



ANCHOR QEA

integral consulting inc.

0 1,000

Feet

Surface Sediment (0-6 Inches) Sample Location

Does Not Exceed REV

Exceeds REV

Vanadium Upstream Reference Envelope Value (REV) = 15.2 mg/kg
Concentrations of the chemical are provided if they exceed one or more criteria

Preliminary Site Perimeter

Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-13
Concentrations of Vanadium in Sediment Relative to the Upstream Reference Envelope Value for Vanadium
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

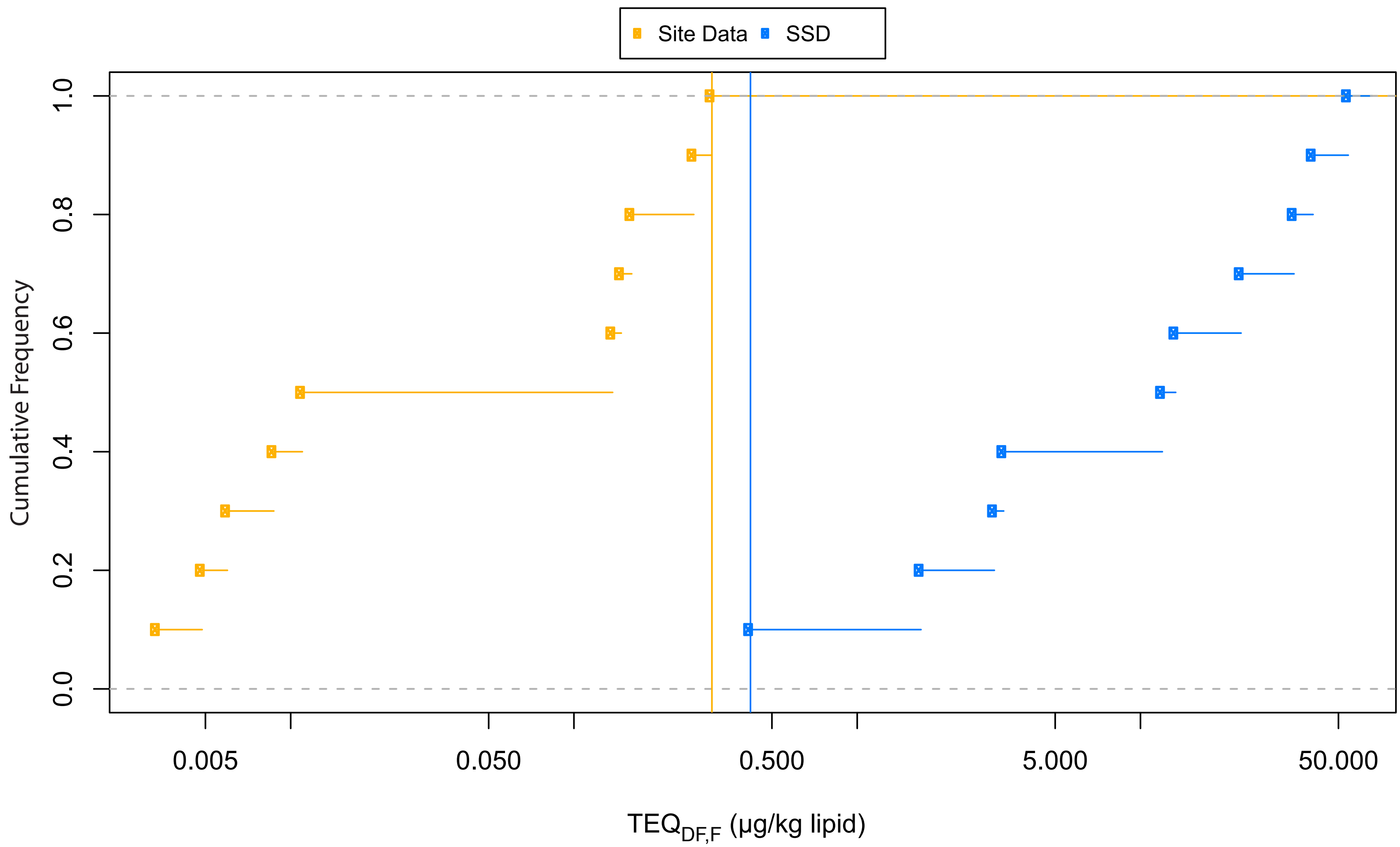
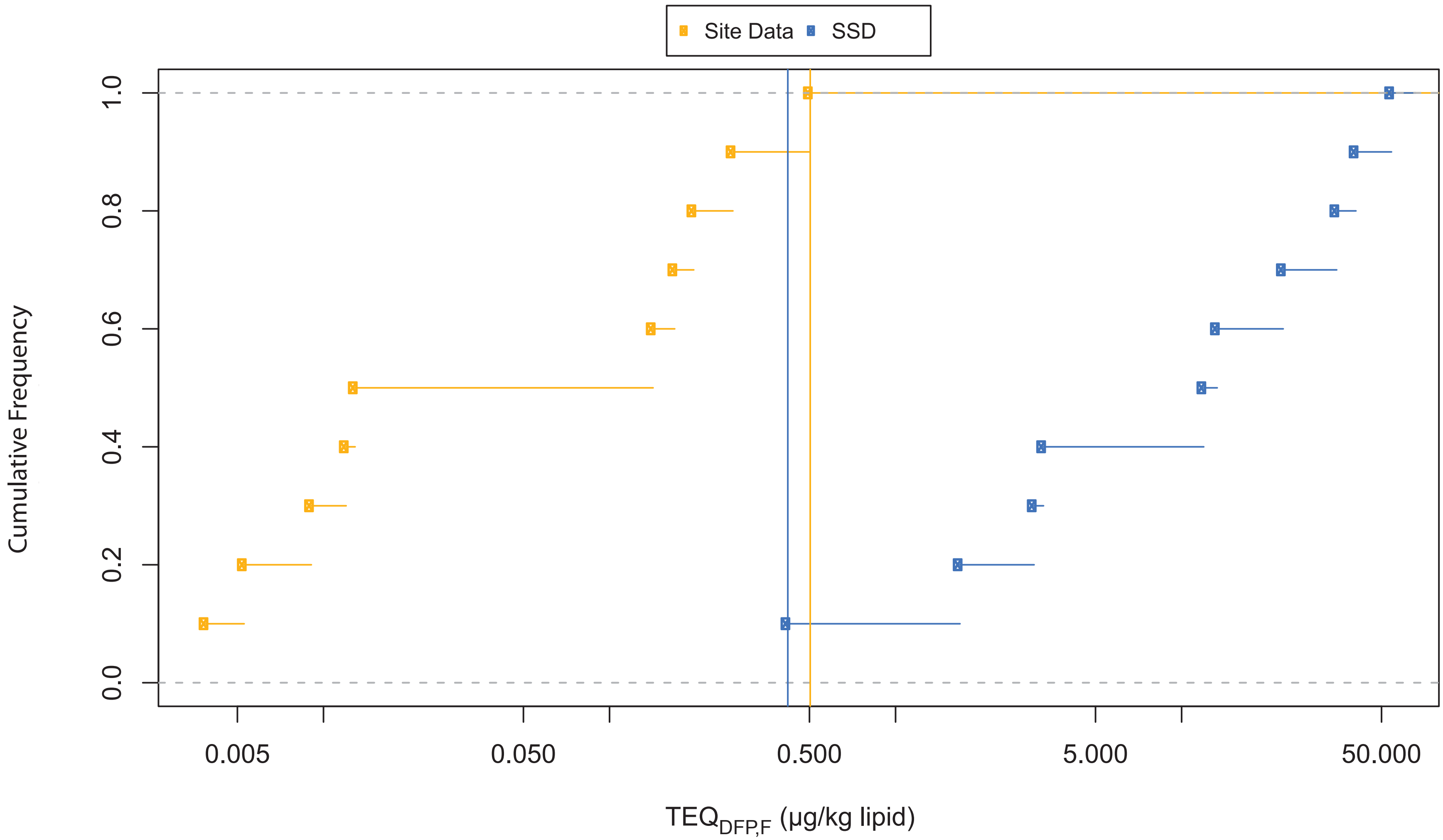
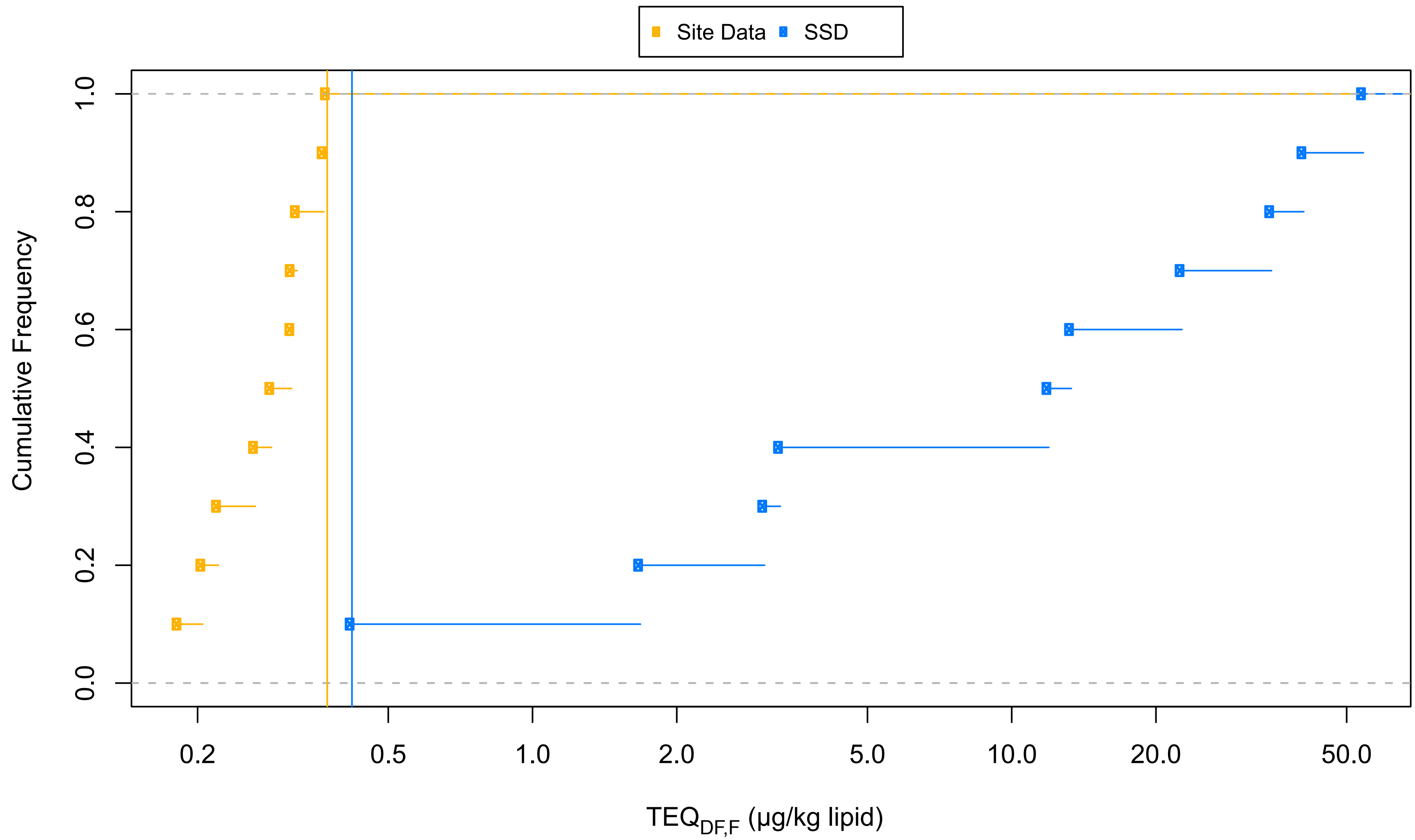


Figure 6-14
Comparison of TEQ$_{DF,F}$ in Gulf Killifish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



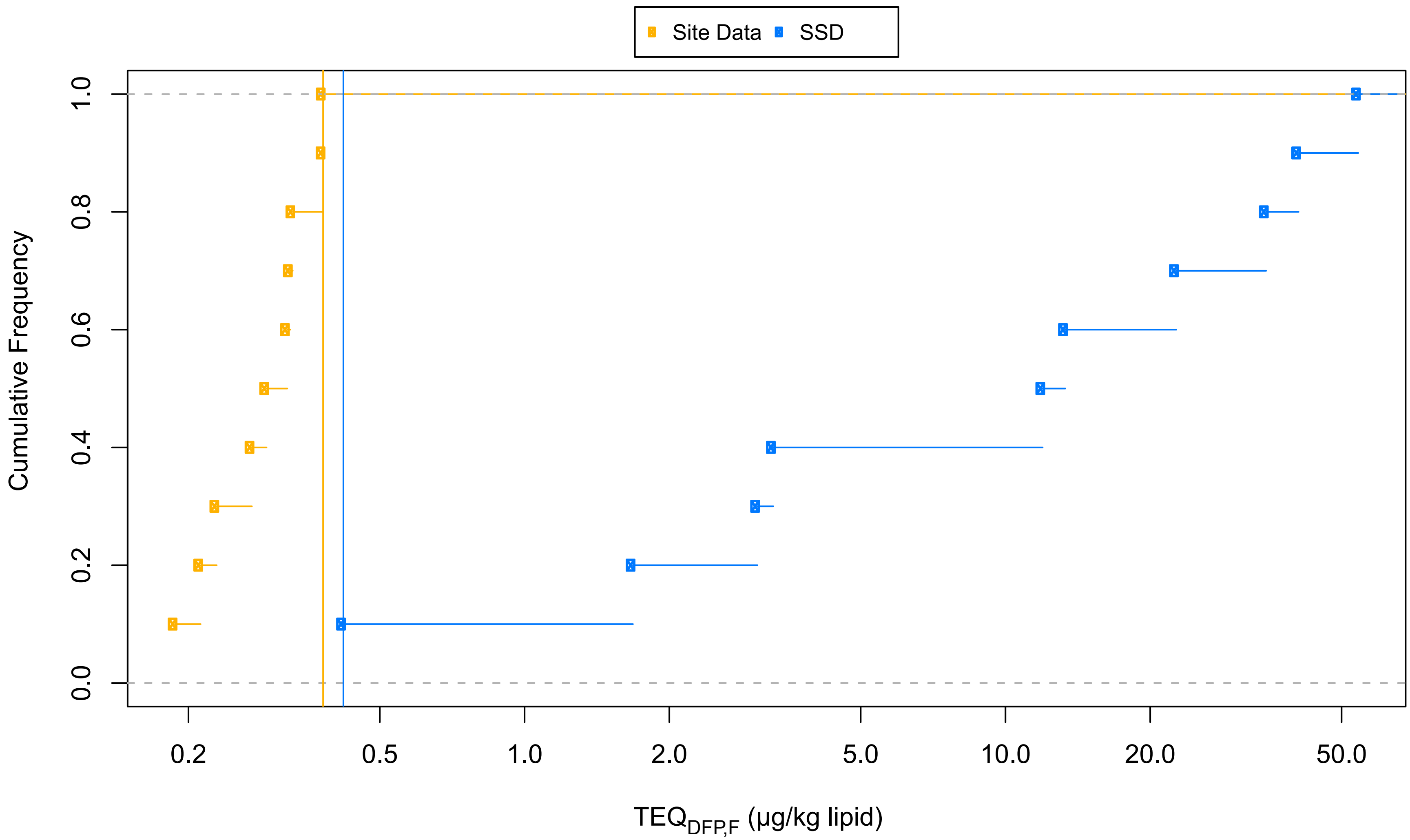
Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 6-15
Comparison of TEQ$_{DFP,F}$ in Gulf Killifish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 6-16
Comparison of TEQ$_{DF,F}$ in Hardhead Catfish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



Note:
Lines adjacent to data points indicate distance to the next data point in the distribution.

Figure 6-17
Comparison of $TEQ_{DFP,F}$ in Hardhead Catfish with the Species Sensitivity Distribution
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

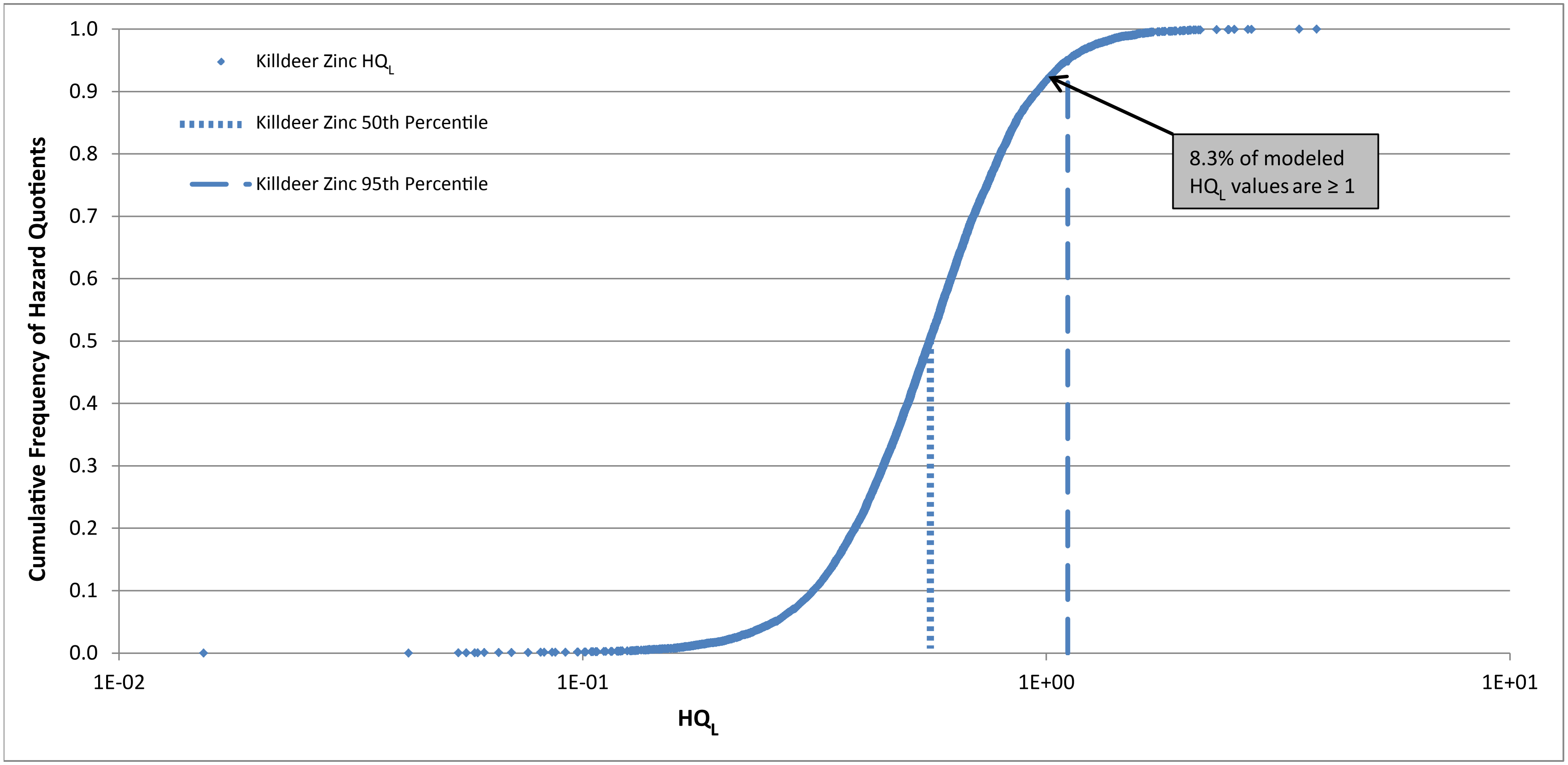


Figure 6-18
Probability Distributions of Zinc Hazard Quotients (HQ_L) for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

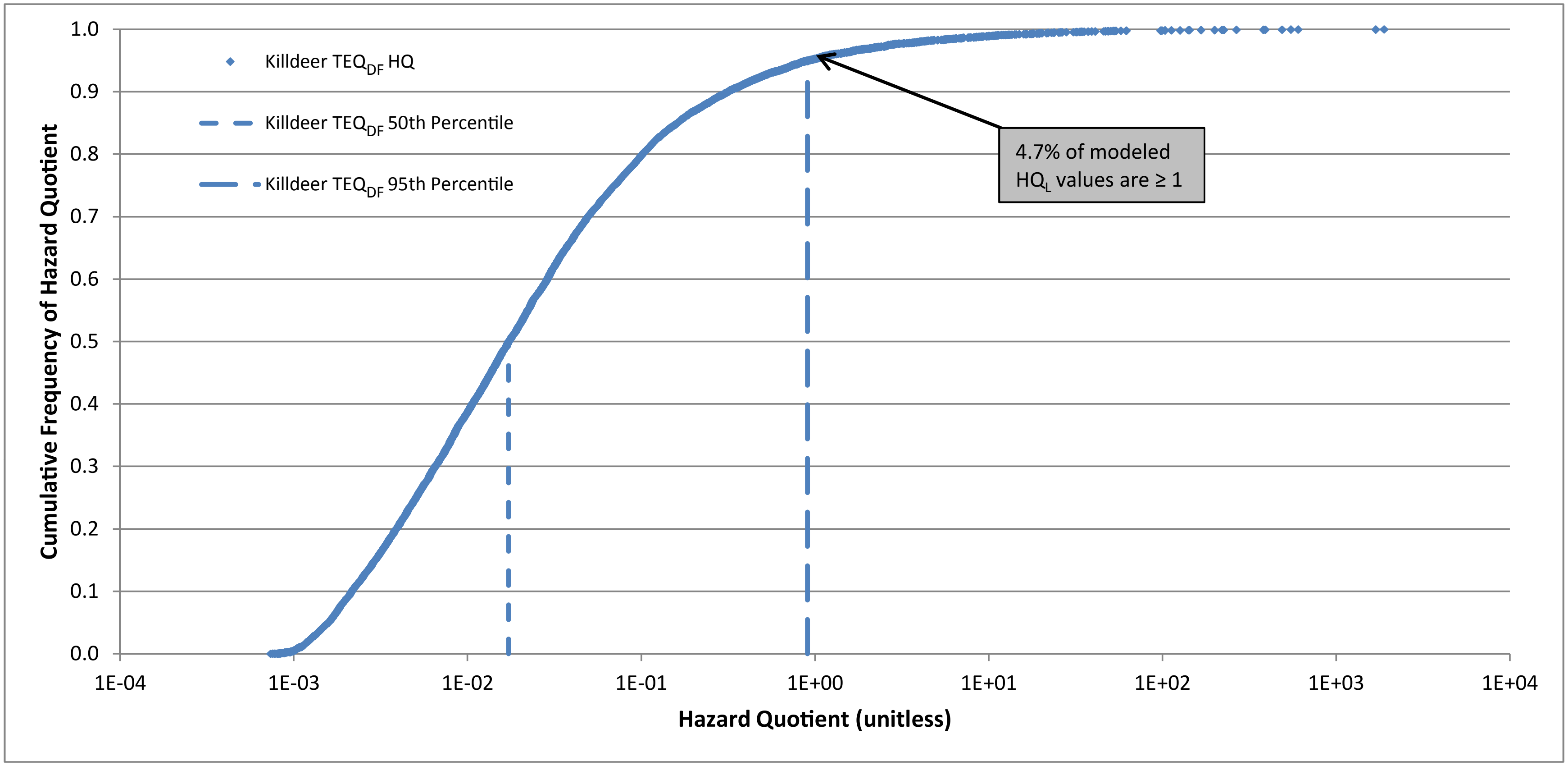


Figure 6-19
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients (HQ_L) for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

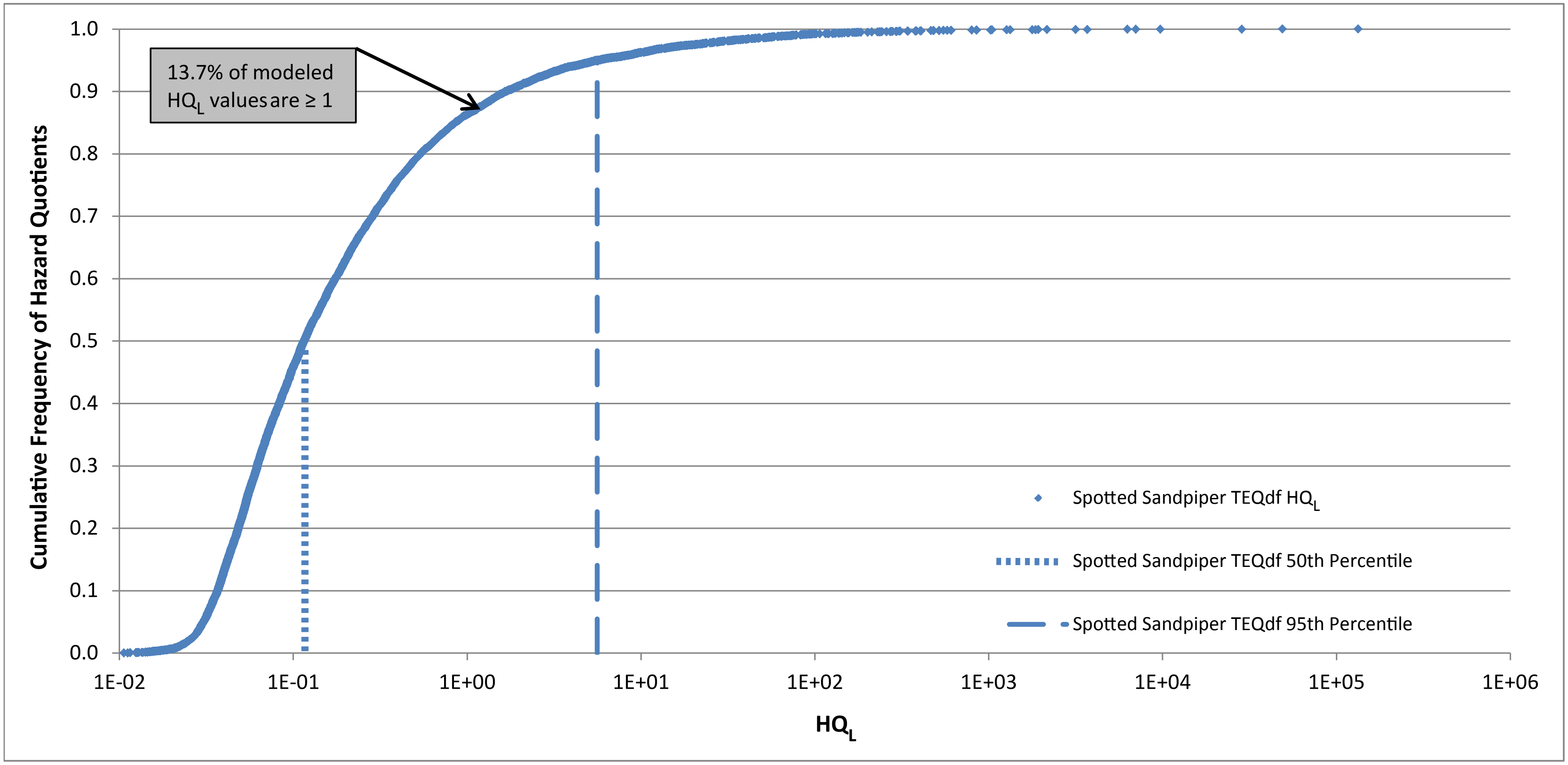


Figure 6-20
Probability Distributions of TEQ$_{DF,B}$
Hazard Quotients for Spotted Sandpiper
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

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ANCHOR QEA
integral consulting inc.

0 1,000
Feet

Surface Soil Sample Location (0 - 6 inches)

- 0.7 - 82.5
- 82.5 - 164.4
- 164.4 - 246.2
- 246.2 - 328
- Preliminary Site Perimeter
- Area Within the Original (1966) Perimeter of the North Impoundments

Figure 6-21
Concentrations of Zinc (mg/kg) in Surface Soils North of I-10
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

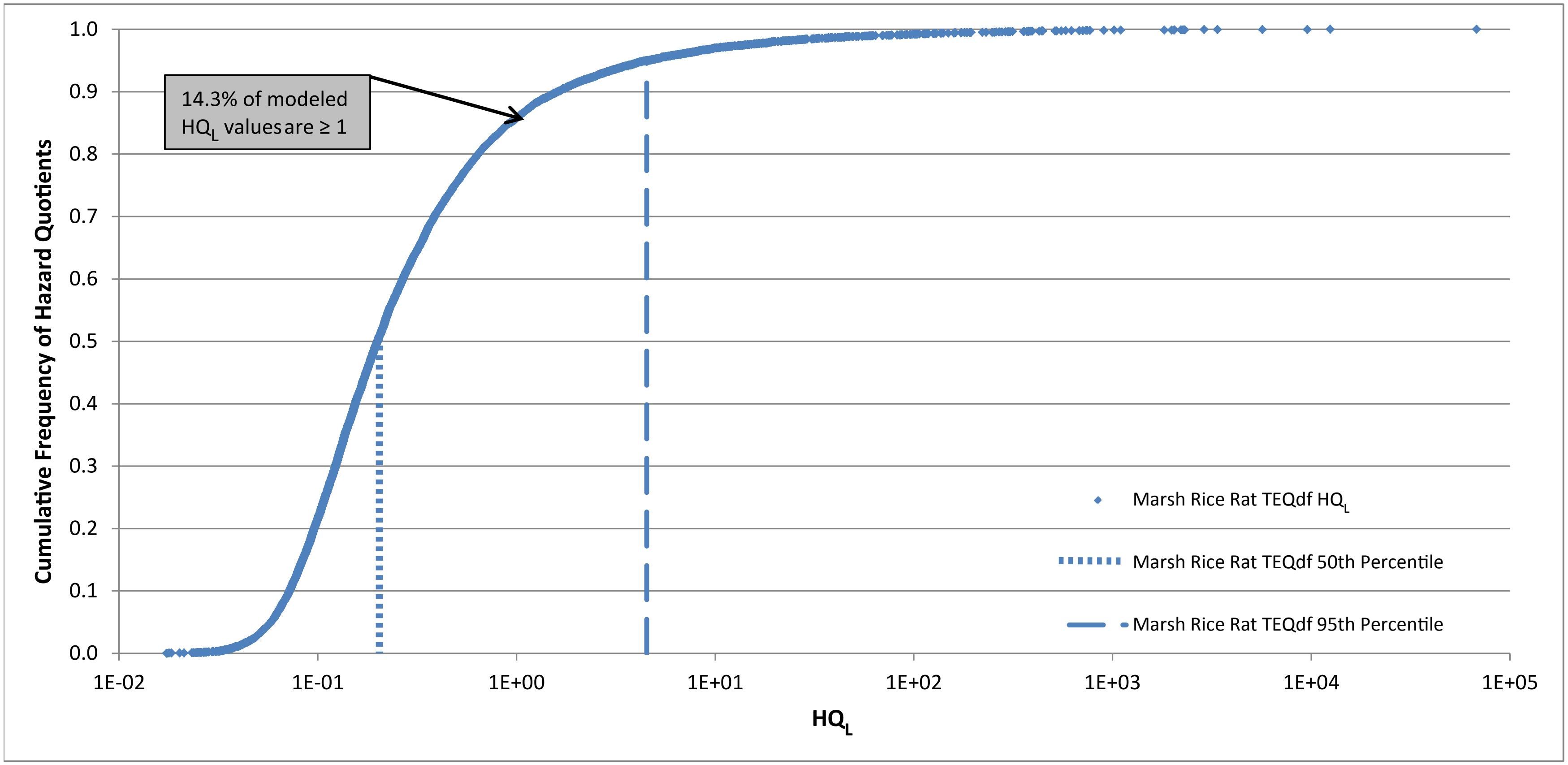

Figure 6-22
Probability Distributions of $TEQ_{DF,M}$ Hazard Quotients for Marsh Rice Rat
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

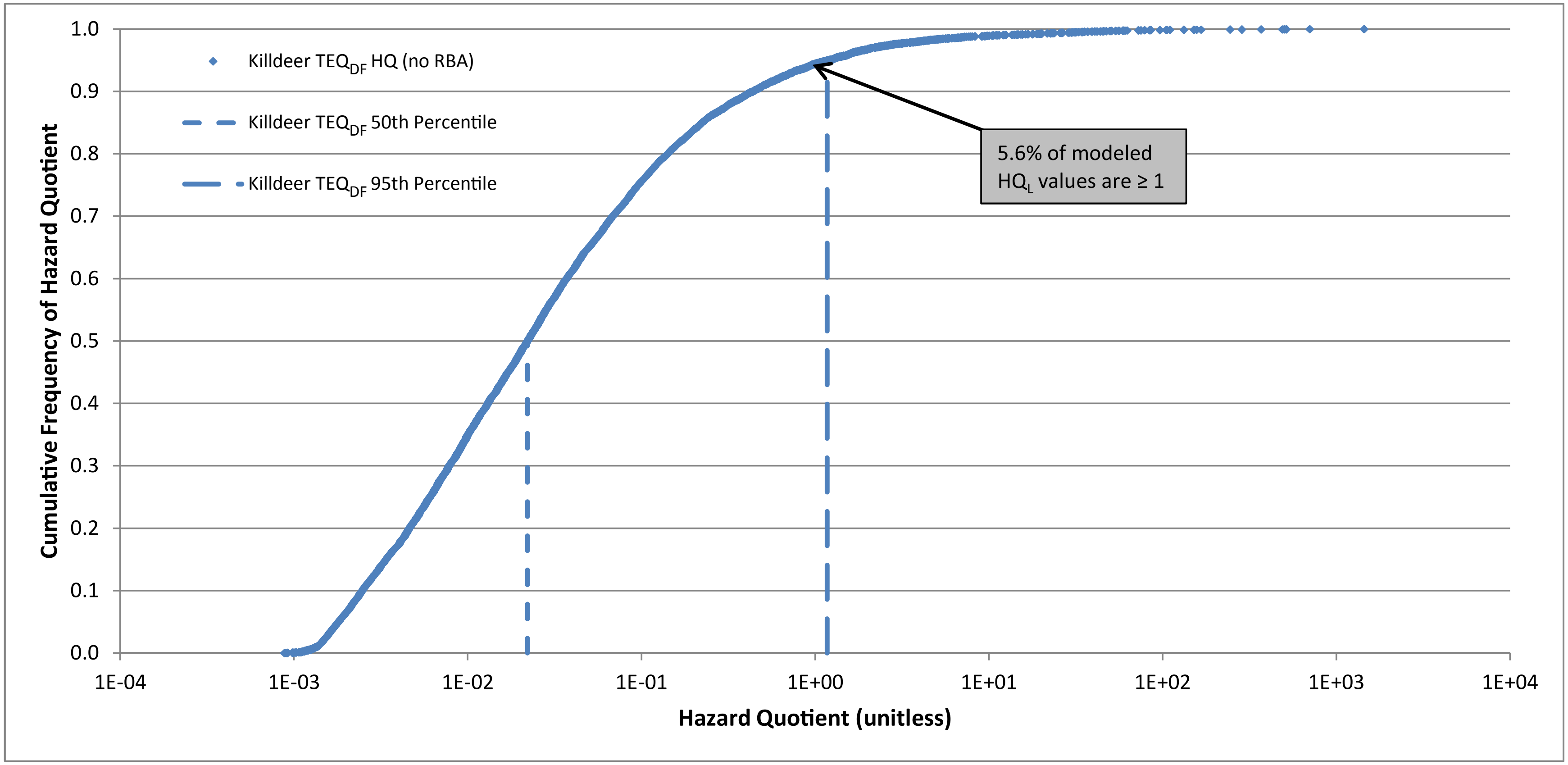

Figure 7-1
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients (HQ_L) with
No Relative Bioavailability Adjustment for Killdeer
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

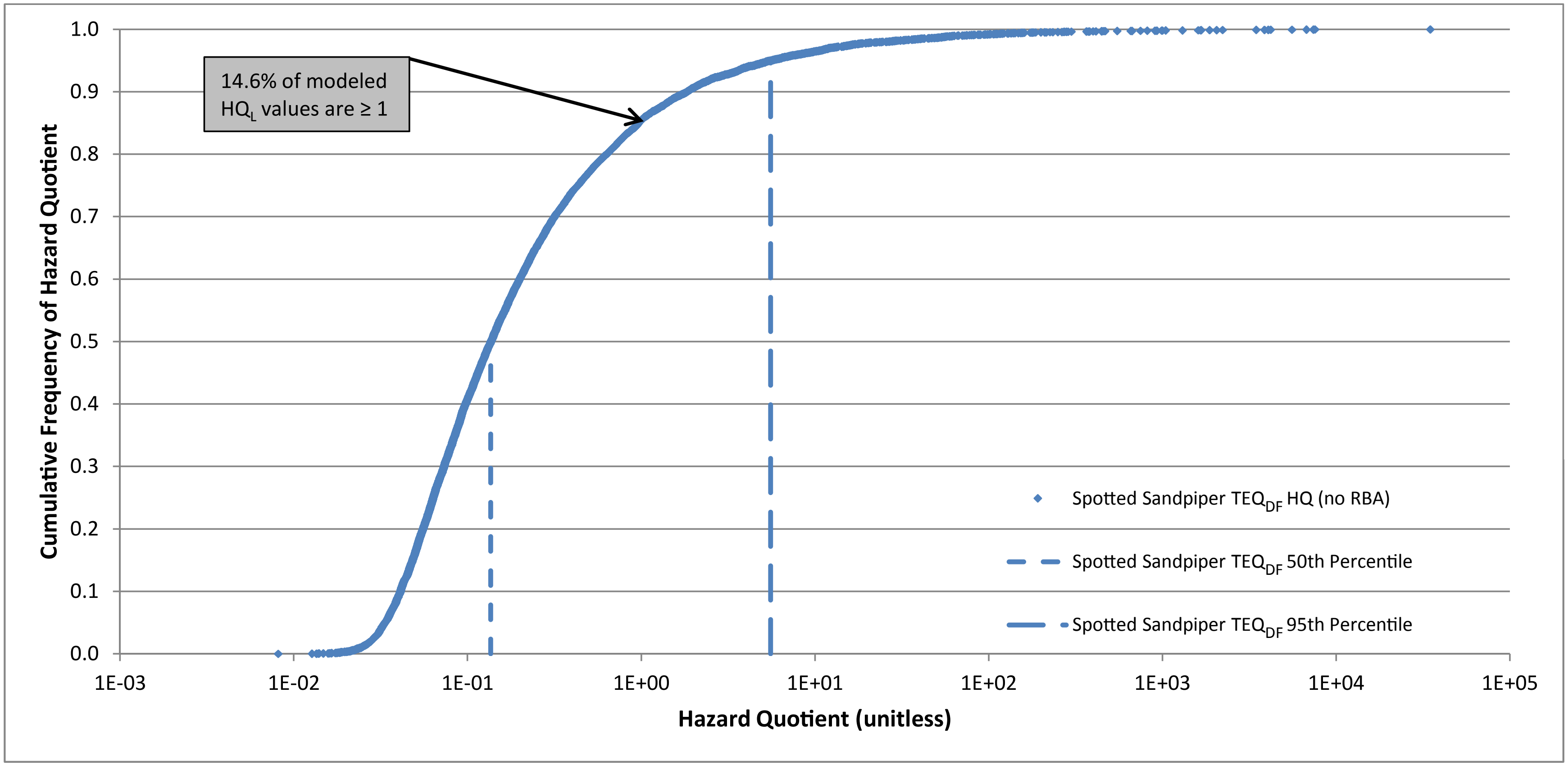

Figure 7-2
Probability Distributions of $TEQ_{DF,B}$ Hazard Quotients (HQ_L) with
No Relative Bioavailability Adjustment for Spotted Sandpiper
Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

EXHIBIT 1
FOOD WEB EXPOSURE MODEL AND EXAMPLE OF ITS APPLICATION FOR SPOTTED SANDPIPER EXPOSURE TO DIOXINS AND FURANS

Exhibit 1

Food Web Exposure Model and Example of Its Application for Spotted Sandpiper Exposure to Dioxins and Furans

This exhibit illustrates how estimates of exposure of site receptors to a given $COPC_E$ were calculated. The example provided in the supporting tables is for spotted sandpiper exposure to dioxins and furans.

To estimate the cumulative daily dose for reptiles, mammals, and birds through ingestion of food and water, including incidental soil or sediment ingestion, the following general equation was used, as described in the BERA:

$$\text{Daily Dose} = \left((\text{FIR} \times C_{\text{food}} \times \text{RBA}_{\text{food}}) + (\text{WIR} \times C_{\text{water}}) + (\text{SIR} \times C_{\text{sed}} \times \text{RBA}_{\text{sed}})\right) \times \text{AUF} \quad \text{(Eq. 4-5)}$$

Where:

Daily Dose	=	$COPC_E$s ingested per day via food, water, and sediment (mg/kg bw day)
FIR	=	food ingestion rate (kg food dw/kg bw day)
C_{food}	=	concentration in the overall diet (mg/kg food dw)
RBA_{food}	=	bioavailable fraction absorbed from ingested prey items (unitless); set to 1 except as described below
WIR	=	water ingestion rate (L water/kg bw day)
C_{water}	=	concentration in water (mg/L water)
SIR	=	sediment ingestion rate (kg sediment dw/kg bw day)
C_{sed}	=	concentration in sediment (mg/kg dw)
RBA_{sed}	=	bioavailable fraction absorbed from ingested sediment or soil (unitless); set to one except as described below
AUF	=	area use factor (unitless); fraction of time that a receptor spends at the site relative to the entire home range

Table 1 provides an example of the parameters described in the above equation for the spotted sandpiper receptor. These parameters are then combined with site data (Table 2) using the equation above to illustrate how an estimated daily dose is calculated for $TEQ_{DF,B}$ to spotted sandpipers at the site (Table 3).

The relative bioavailability adjustment factors, RBA_s and RBA_{food}, are both set to equal 1, assuming complete bioavailability, except for the dioxin congener 2,3,7,8-TCDD. For this congener, an RBA_{sed} of 0.41 and RBA_{food} of 0.44, for invertebrate food items only, as described in Section 4.3.1.2 of the BERA, are applied to concentrations of 2,3,7,8-TCDD in the database prior to calculation of the $TEQ_{DF,B}$.

Exhibit 1

Food Web Exposure Model and Example of Its Application for Spotted Sandpiper Exposure to Dioxins and Furans

Table 1. Estimated Daily Dose of $TEQ_{DF,B}$[a] to Spotted Sandpiper

	Dose from Food	**Dose from Water**	**Dose from Sediment**	**Total Dose**
Equation	FIR x ΣC_{food}[b]	WIR x C_{water}	FIR x F_{sed} x C_{sed}	[(FIR x ΣC_{food}) + (WIR x C_{water}) + (FIR x F_{sed} x C_{sed})] x AUF
Units	mg/kg bw-day	mg/kg bw-day	mg/kg bw-day	mg/kg bw-day
$TEQ_{DF,B}$	1.12E-05	4.40E-09	1.36E-04	1.47E-04

[a]Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.

[b]$\Sigma C_{food} = \Sigma (C_{food1} \times F_{food1} + C_{food2} \times F_{food2} ... C_{foodn} \times F_{foodn})$, where F_{food} is the fraction of food in the diet

Table 2. Concentrations in Ingested Media Used to Calculate the Daily Dose to Spotted Sandpiper in Table 1

	Concentrations in Ingested Media			
Exposure Area	Site-wide Aquatic	Shoreline	Site-wide Aquatic	
Equation	C_{water}	C_{sed}	$C_{food\ -\ crustacea}$	$C_{food\ -\ molluscs}$
Units	mg/L	mg/kg	mg/kg	mg/kg
$TEQ_{DF,B}$[a]	2.63E-08	3.43E-03	1.95E-05	8.22E-05

[a]Values shown are for the central tendency exposure.

Table 3. Spotted Sandpiper Life History Parameters Used to Calculate the Daily Dose in Table 1

Component	**FIR**	$F_{food\text{-}crustacea}$	$F_{food\text{-}molluscs}$	F_{sed}	**WIR**	**AUF**
Description	Food Ingestion Rate	Crustacea Fraction of Total Diet	Molluscs Fraction of Total Diet	Fraction of Soil or Sediment in Diet	Water Ingestion Rate	Area Use Factor
Units	kg food/kg bw-day	kg food/kg diet	kg food/kg diet	kg sediment/kg diet	L water/kg bw-day	km habitat/km home range
Value	0.22	0.5	0.5	0.18	0.17	1

EXHIBIT 2A
EXAMPLE CALCULATIONS FOR ESTIMATION OF $TEQ_{DF,B}$ CONCENTRATIONS IN BIRD EGGS

Exhibit 2A

Example Calculations for Estimation of $TEQ_{DF,B}$ Concentrations in Bird Eggs

Each case uses the linear regression equation taken from Elliott et al. 2001,

log(Egg Conc.) = a X log(Ingested Conc.) + b

Where: Egg Conc. = estimated concentration in egg (ng/kg ww)
Ingested Conc. = Concentration (ng/kg ww) of each congener calculated by summation of the concentrations of that congener from each dietary source scaled by its respective fractional contribution to the total mass of ingested media.
a = slope (as determined by Elliott et al. 2001 and applied in this estimate; Table 4-14)
b = intercept (as determined by Elliott et al. 2001 and applied in this estimate; Table 4-14)

Predicted TEQ concentrations were determined for the range of TEFs in each congener group (Table 4-16).

Case 1: Calculation of estimated $TEQ_{DF,B}$ egg concentration in cormorant consuming prey only. Values represent the CT of $TEQ_{DF,B}$ ingested prey, and TEQ was calculated using the maximum TEF.

Formula Used in Cases 1-3:

$$\left[\left(\text{Conc. in Media (ng/kg)} \times \text{Fraction of Diet}\right) = \text{Fraction of Ingested Conc.}\right] \text{Ingested Conc.} \rightarrow \left[\left(\text{log(Ingested Conc.)} \times a\right) + b\right] = \text{log(Egg Conc.)} \rightarrow \left(\text{Estimated Conc. in Egg} \times \text{TEF}\right) = \text{Predicted TEQ Conc.}$$

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf Killifish	TCDF	GK-TTR5-2	0.618	1.00	0.618	0.618	-0.209	0.407	0.333	0.248	1.77	1	1.77
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Case 2: Calculation of estimated $TEQ_{DF,B}$ egg concentration for PeCDD in heron consuming three prey types and shoreline sediment. Values used represent the RM of $TEQ_{DF,B}$ in each ingested medium. TEQ concentration is calculated using maximum TEF value for the PeCDD group.

Bird Receptor	Ingested item for scenario involving consumption of prey and sediment	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Heron	Blue crab	PeCDD	SJFCA1-CR6	1.16	0.010	0.012	0.470	-0.328	0.647	1.832	1.620	41.7	1	41.7
	Gulf killifish		GK-TTR3-2	0.00995	0.495	0.005								
	Hardhead catfish		SJFCA1-LF1	0.0236	0.495	0.012								
	Shoreline sediment		SJNE022-2	13.4	0.033	0.442								
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Case 3: Calculation of estimated $TEQ_{DF,B}$ egg concentration for HxCDF in sandpipers consuming two prey types and shoreline sediment. Values used represent the CT of $TEQ_{DF,B}$ in each ingested medium. TEQ concentrations is calculated using minimum TEF value for the HxCDF group.

Bird Receptor	Ingested item for scenario involving consumption of prey and shoreline sediment	Congener	Sample ID	Conc. in Media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	log(Ingested Conc.)	a	b	log(Egg Conc.)	Estimated Conc. In Egg	TEF	Predicted TEQ Conc.
Sandpiper	Blue crab	HxCDF	SJFCA2-CR6	0.087	0.50	0.044	30.650	1.486	0.741	1.400	2.501	317	0.1	31.7
	Common rangia		CL-TTR5-001	0.012	0.50	0.006								
	Shoreline sediment		TCEQ2009_03	170	0.18	30.6								
			Table 4-15		Table 3-12				Table 4-14				Table 4-16	Table 4-17

Notes

CT = central tendency
RM = reasonable maximum
TEF = toxicity equivalence factor
$TEQ_{DF,B}$ = toxicity equivalent for dioxins and furans calculated using avian toxicity equivalence factors with nondetects set at one-half the detection limit

EXHIBIT 2B
EXAMPLE CALCULATIONS FOR ESTIMATION OF PCB CONCENTRATIONS IN BIRD EGGS

Exhibit 2B
Example Calculations for Estimation of PCB Concentrations in Bird Eggs

Egg Conc. = Ingested Conc. X BMF

Where: Egg Conc. = estimated concentration in egg (ng/kg ww)
Ingested Conc. = Concentration (ng/kg ww) of congener calculated by summation of the individual congener concentrations from each dietary source scaled by its respective fractional contribution.
BMF = biomagnification factor for each PCB from ingested medium to egg (Table 4-18)

Predicted TEQ concentrations were determined by application of congener specific TEF (Table 4-18).

Case 1: Calculation of estimated $TEQ_{P,B}$ egg concentration in cormorant consuming prey and sediment for PCB105. Values represent the CT of $TEQ_{P,B}$ in prey and sediment samples.

Formula Used in Cases 1-4: (Conc. in Media (ng/kg) X Fraction of Diet) = Fraction of Ingested Conc. → (Ingested Conc. X BMF) → Estimated Conc. in Egg X TEF = Predicted TEQ Conc.

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf killifish	PCB105	GK-TTR5-1	715	1.00	715	819	20	16,372	0.0001	1.64
	Sediment		SJNE022-1	5,180	0.02	104					

Case 2: Calculation of estimated background $TEQ_{P,B}$ egg concentration in cormorant consuming prey and sediment for PCB126. Values represent the RM of $TEQ_{P,B}$ in prey and sediment samples.

Bird Receptor	Ingested item for scenario involving consumption of prey only	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Cormorant	Gulf killifish	PCB126	GK-TTR7-1	5.43	1.00	5.430	5.46	18.7	102	0.1	10.2
	Sediment		SJNE065	1.64	0.02	0.033					

Case 3: Calculation of estimated background $TEQ_{P,B}$ egg concentration in heron consuming prey for PCB077. Values represent the RM of $TEQ_{P,B}$ for prey samples.

Bird Receptor	Ingested item for scenario involving consumption of prey and sediment	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Heron	Blue crab	PCB077	SJFCACB-CR1	9.06	0.010	0.091	18.7	0.7	13.1	0.05	0.655
	Gulf killifish		GK-TTR7-1	4.42	0.495	2.19					
	Hardhead catfish		SJFCACB-LF6	33.2	0.495	16.4					

Exhibit 2B
Example Calculations for Estimation of PCB Concentrations in Bird Eggs

Case 4: Calculation of estimated $TEQ_{P,B}$ egg concentration in sandpiper consuming prey for PCB118. Values represent the CT of $TEQ_{P,B}$ for prey samples.

Bird Receptor	Ingested item for scenario involving consumption of prey and shoreline sediment	Congener	Sample ID	Conc. in media (ng/kg)	Fraction of Diet	Fraction of Ingested Conc.	Ingested Conc.	BMF	Estimated Conc. in Egg	TEF	Predicted TEQ Conc.
Sandpiper	Blue crab	PCB118	SJFCA2-CR6	2,574	0.50	1,287	2,687	4.61	12,386	0.00001	0.124
	Common rangia		CL-TTR3-002	2,800	0.50	1,400					

APPENDIX A
RECEPTOR PROFILES

RECEPTOR PROFILES
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
Site	San Jacinto River Waste Pits Superfund site
USEPA	U.S. Environmental Protection Agency

1 INTRODUCTION

Receptors were selected for the baseline ecological risk assessment (BERA) to represent species of potential concern and the range of feeding guilds expected to inhabit terrestrial and/or aquatic habitat types at the San Jacinto River Waste Pits Superfund site (Site). Patterns of behavior, life history, and habitat use that affect the frequency and magnitude of exposure of each receptor to hazardous substances at the Site are quantified to estimate exposures for the ecological risk assessment. Quantitative estimates of parameters describing rates of ingestion of soil and food; information on life history (e.g., timing of migration and breeding); and habitat areas for each receptor are used to build exposure models. Specific information about how receptors use habitats at the Site can also be used to interpret the ecological significance of estimated exposures relative to effects thresholds.

Receptor surrogates for evaluation in the BERA for the northern impoundment were selected in the screening level ecological risk assessment (Appendix B to the Remedial Investigation and Feasibility Study Work Plan) as follows:

- Benthic macroinvertebrate communities
- Fish
  - Gulf killifish (*Fundulus grandis*)
  - Black drum (*Pogonias cromis*)
  - Southern flounder (*Paralichthys lethostigma*)
- Reptiles
  - Alligator snapping turtle (*Macrochelys temminckii*)
- Piscivorous birds
  - Neotropic cormorant (*Phalacrocorax brasilianus*)
  - Great blue heron (*Ardea herodias*)
- Invertivorous birds
  - Spotted sandpiper (*Actitis macularius*)
  - Killdeer (*Charadrius vociferous*)

- Semiaquatic mammals
  - Raccoon (*Procyon lotor*)
  - Marsh rice rat (*Oryzomys palustris*).

Aquatic and terrestrial invertebrates are not discussed in this Appendix. Only fish, reptiles, birds, and mammals are discussed below because specific life history or biological factors affect exposures in a manner relevant to risk estimation for these receptors. Information on the proposed receptors for the south impoundment is also discussed herein (killdeer, pocket gopher [*Geomys breviceps*], common garter snake [*Thamnophis sirtalis*]; see also Appendix E) and will be used in the BERA for the south impoundment. Quantitative biological variables required for modeling the exposures of fish, alligator snapping turtle, birds, and mammals included:

- Body weight (kg)
- Average home range (km or km^2)
- Rates of ingestion of food, water (for reptiles, seabirds, and wading birds) and soil or sediment (kg diet/kg body weight [bw]-day)
- Composition of the diet.

Because surface water in the vicinity of the Site is brackish, wildlife other than reptiles, seabirds, and wading birds are not expected to ingest water at the Site, so water ingestion is not modeled as part of the exposure estimation for mammals or terrestrial birds. However, seabirds such as cormorants have nasal glands that allow them to concentrate and excrete salts from the blood following ingestion of saline waters, while wading birds like sandpipers have been shown to tolerate ingestion of water at low salinities (Purdue and Haines 1977) such as are present at the site.

Other variables that can be important to make risk models more realistic or that support qualitative interpretation of estimated exposures relative to effects levels include:

- Numbers of breeding cycles per year
- Numbers of young per brood and period of gestation
- Seasonal patterns of migration and months expected on site
- Seasonal changes in diet

- Preferences for certain habitat types or physical conditions
- Density and territoriality.

To compile the necessary information, relevant descriptors of each receptor were taken from primary scientific publications, from synthetic reviews *(e.g., Wildlife Exposure Factors Handbook*; USEPA 1993), and from ecological risk assessments conducted in U.S. Environmental Protection Agency (USEPA) Region 6 (e.g., Patrick Bayou Superfund Site). Available data include a range of studies from across North America and Europe conducted in a variety of climates and habitat types. In many cases, only one or two studies were available to describe a species of interest. When multiple values for a relevant exposure parameter were found in the literature, the following were considered in the final selection of a quantitative value:

- Geographic proximity to the Site
- Ecological similarity to the Site (e.g., southern temperate estuarine and mixed-deciduous riparian systems)
- Climatic similarity to the Site (e.g., southern temperate ecosystems).

The following sections provide the basis for specific exposure assumptions used in the exposure modeling. Detailed exposure assumptions are presented in Section 4 of the main text of the report. Receptor profiles also provide context for interpretation of risk models.

2 FISH

Three fish species found at the Site are used as receptor surrogates for the ecological risk evaluation: Gulf killifish, black drum, and southern flounder. These species were selected because they incorporate several life history characteristics important to measuring exposure to bioaccumulative contaminants, including long lifespans, limited home ranges (a focus on non-migratory species), proximity to sediments and feeding from the benthic environment, and mid- and upper-trophic level diets. They also represent important prey species for upper trophic levels. Relevant information on each species is summarized below.

2.1 Gulf Killifish

The Gulf killifish is a relatively small (up to 18 cm), omnivorous, euryhaline fish that is commonly found in estuaries, tidal marshes, bay shores and river reaches of the Gulf Coast. Native to the western Gulf of Mexico slope drainages, the Gulf killifish, often called the "bait minnow" in Texas, inhabits drainages along the coast from the northeastern coast of Florida to the Gulf Coast of Texas and Cuba (Schofield and Fuller 2011; Hassan-Williams et al. 2011). While these fish tolerate a wide range of salinities, they are reported to favor open, brackish waters to inland ponds (Hassan-Williams et al. 2011). The Gulf killifish is among the most abundant fish species in vegetated areas (inner *Spartina alterniflora* or *S. patens* marsh) of upper Galveston Bay in the spring and fall (Rozas and Zimmerman 2000, as cited in Hassan-Williams et al. 2011).

The Gulf killifish feeds throughout the water column, preying on grass shrimp (*Palaemonetes*), microcrustaceans (copepods), mosquito (*Dipteran*) larvae and pupae, and small fishes, terrestrial insects on the surface, and bivalve molluscs, benthic algae and aquatic plants (Schofield and Fuller 2011; Hassan-Williams et al. 2011). They have been observed to increase their feeding rate when gaining access to flooded marshes (Schofield and Fuller 2011).

In Texas, the spawning season for Gulf killifish is from March to October. The preferred habitat for spawning is shallow areas within dense beds of marsh grass, clumps of debris, or among oysters. Eggs are deposited on vegetation during periods of maximum high tides (spring tides), develop while they are exposed to humid air, and hatch when inundated on the next extreme high tide, usually in about 11 to 15 days (Schofield and Fuller 2011; Hassan-Williams et al. 2011). Gulf killifish are non-migratory.

2.2 Black Drum

The black drum is a large-bodied, estuarine invertivore, found commonly in the bays and estuaries of the Gulf of Mexico. This fish is a member of the croaker family, so named because of its ability to produce croaking or drumming sounds with its air bladders. Younger, smaller fish, under a pound in weight, are referred to as "butterfly drum," while older drum of 30 pounds or more, of either gender, are called "bull drum." Black drum are widely distributed and abundant throughout Gulf waters, including bay, inshore, and offshore areas, and are among the most commonly caught sport fish and a mainstay in the commercial fishing industry (TPWD 2011a).

The black drum is found along the western Atlantic Coast from Nova Scotia south through the Gulf states to Mexico and along the southern Caribbean coast. It is a versatile species that can adapt to a wide range of habitats, from extremely warm clear waters found in shallow flats during the summer to Gulf waters at depths of more than 100 feet, and are known to survive better than many fish in freezing conditions (TPWD 2011a; Froese and Pauly 2009). They are attracted to freshwater creeks and rivers, can tolerate very salty waters, and have been found in turbid and muddy flooding sloughs.

The diet of the black drum consists primarily of molluscs, crab, and shrimp (TPWD 2011a). Black drum are very adaptable and thus are found year round in a very wide variety of estuarine and marine habitats along the Gulf Coast. Black drum are resident within localized embayments when forage is plentiful and water conditions are acceptable, but have been observed to migrate in search of food and more desirable habitats, as far as 245 miles in 1 year, though distances of less than 10 miles are more common. Spawning migrations are notable. Large drum, at least 4 to 5 years of age, school in deeper bays and channels before spawning, an event known as a "Bull Run" among anglers. Free spawning occurs in these bays and channels in Texas primarily in February and March, and in the Gulf through April, with some spawning occurring later, in June and July (TPWD 2011a; Hill 2005).

Larval black drum inhabit the surf and bay shorelines in March and April and the half-inch to inch-long juveniles are found in early summer in shallow muddy creeks, sloughs, and areas that provide structure and cover (Froese and Pauly 2009). Young drum reach about 6 inches in length in the first year, 12 inches by the second year, and about 16 inches by the third year (TPWD 2011a). Black drum continue to grow and gain about 2 inches per year until they reach about 20 years of age. Females, which are characterized as prodigious, multiple spawners, mature at age 4 to 6 years (FWRI 2010). This species has been known to live to almost 60 years (FWRI 2010). Most bull drum weigh between 30 and 45 pounds; the largest black drum taken in Texas by a sports angler weighed 78 pounds (TPWD 2011a). Juvenile black drum are prey to a wide range of estuarine piscivores, including spotted seatrout (*Cynoscion nebulosus*) and crevalle jack (*Caranx hippos*). Larger drum are eaten by sharks (FWRI 2010).

2.3 Southern Flounder

Southern flounder is a large piscivorous flatfish common in the Gulf of Mexico. It is euryhaline, demersal (living and feeding near the bottom), cryptic (camouflaged) and resides over mud bottoms in estuaries and in coastal waters up to 40 m in depth (Froese and Pauly 2009). The southern flounder is found in the western Atlantic, ranging from North Carolina to the Texas coast and southward into Mexico, though is absent from the southern Florida coast and usually found west of the Mississippi River (Froese and Pauly 2009; TPWD 2011b). Mature southern flounder migrate out to the open Gulf in the winter to spawn and return to the estuaries in early spring, where they reside within their chosen home range for the remainder of the year. This species, which is a highly prized commercial food fish, is also targeted by recreational anglers at inshore bridges, jetties and from small boats in marshes and tidal creeks (Froese and Pauly 2009). The southern flounder tolerates low salinities and is found frequently in brackish bays and estuaries, and even on occasion in fresh water (Froese and Pauly 2009), except in the winter when most adult flounder have moved out into the Gulf of Mexico to spawn.

As with other flounder species, the mature southern flounder is compressed laterally, with the dorsal side of the body pigmented, the underside pale, and both eyes present on the dorsal (left) side of the body. The flounder is well-adapted to its habitat in that both eyes in adults are on the “up” side of the head and the fish can alter the pigmentation of the upper side of the body to match the surrounding environment. The absence of an air bladder, as well as its small body cavity, help the flounder stay close to, or lie on the bottom and wait to ambush prey (TPWD 2011b).

Juvenile flounder feed mostly on small benthic invertebrates such as crustaceans, and add fish increasingly to their diet as they grow (Hill 2005). Adult southern flounders are almost strictly piscivorous, but will opportunistically feed on large invertebrates (i.e., crabs and shrimp) (Hill 2005).

Female southern flounder reach maturity at about 2 years and, coincidental with a 4 to 5°C drop in water temperature in the fall, leave the bays for spawning in the Gulf of Mexico, trailing mature males who move a few weeks earlier (Hill 2005). In Texas, this migration occurs primarily between October and December. Spawning occurs at depths of 50 to 100 feet. The flounder eggs are buoyant and hatch within the Gulf. By late winter and early spring, the larval fish, whose eyes are located on opposite sides of the head and who swim in an upright position are passively transported by tides and currents through Gulf passes into tidal estuaries and flats, where they settle to silty or muddy substrates to undergo metamorphosis. During metamorphosis, the right eye “migrates” to the left side of the head and the fish ultimately assumes its left-side-up position for life (Hill 2005; TPWD 2011b). Juvenile flounder then enter lower salinity waters of coastal rivers, creeks, and bayous. Small flounder grow rapidly and may reach 12 inches in length by the end of their first year (TPWD 2011b). Females grow faster than do males, reaching an average of 12 to 14 inches by maturity. Males average from 8 to 12 inches in length (SCDNR 2011). Adult southern flounders can grow as long as three feet and weight up to 20 pounds.

3 REPTILES

The alligator snapping turtle and the common garter snake were selected as reptile receptors for the BERA. Although it is expected to be very rare in the brackish waters such as those in the San Jacinto River estuary, the alligator snapping turtle was selected as a receptor surrogate because it spends most of its life in water and is omnivorous, consuming plants, small fish, insects, snakes, and carrion. The garter snake was selected because it is a common, invertivorous reptile whose habitat requirements overlap with the conditions present in the upland portions of the site, and it is used for evaluating exposure to reptiles in the south impoundment (Appendix E). Additional information on these receptors' life histories and feeding behaviors is provided below.

3.1 Alligator Snapping Turtle

The alligator snapping turtle, which is the largest freshwater turtle species in the world, resides in freshwater areas of the southeastern portion of the United States, ranging from northern Florida and southern Georgia through the Mississippi River Valley and Gulf states into eastern Texas and Oklahoma (Nichols et al. 1999). The most established populations of this turtle are found around large water bodies, such as the Mississippi River, but they have been found in a variety of environs including lakes, oxbows, bayous, deep rivers, canals, creeks, ponds and brackish estuaries (Franklin 2011). Although primarily a freshwater species, alligator snapping turtles may occasionally use brackish habitats; for example, an adult male was collected in upper Mobile Bay, Alabama, with brackish-water barnacles on its carapace, indicating it had spent sufficient time in saline waters for these organisms to attach to and grow on the carapace (Jackson and Ross 1971). Adult alligator snapping turtles are characterized by three large, pronounced ridges that run from the front to the back of their carapace. Like all snapper turtles, the alligator snapper head is large, its jaws are very powerful, and the turtle is unable to retract the head into the shells. While some captive specimens have reached very large proportions, males in the wild average 26 inches in shell length and weigh about 175 pounds and females are much smaller, with a maximum recorded weight of around 50 pounds (National Geographic 2011). Alligator snappers can live in the wild for 50 to 100 years, though the average life spans for males and females of this species are estimated to be 26 and 23 years, respectively (Nichols et al. 1999).

Alligator snapping turtles are opportunistic omnivores, feeding primarily on fish and other turtles, and less frequently on frogs, snakes, snails, worms, clams, crayfish, carrion, insects and occasionally aquatic plants. These reptiles spend most of their time in the water, crawling on the bottoms of the deepest areas within large rivers, canals, lakes, and swamps during the day; they are known to stay submerged for 40 to 50 minutes, coming to the surface only for air (Nichols et al. 1999; Fuller and Somma 2011). Turtles thermoregulate, using different water depths, seasonally to retain optimal body temperature. Daytime feeding in the depths occurs mainly through ambush of prey (Fuller and Somma 2011). These turtles are more active at night, when they hunt and scavenge the shorelines. Only the nesting female ventures on land, and can feed on smaller terrestrial mammals such as squirrels, muskrats (*Ondatra zibethicus*), nutria (*Myocastor coypus*), and even opossums (family *Didelphimorphia*) and raccoons (Nichols et al. 1999). The only known predators of adult alligator snapping turtles are humans. The eggs and hatchlings of this species are an important source of food for large fish, raccoons, and birds (Nichols et al., 1999).

Alligator snapping turtles mate in early spring in Florida and late spring in the Mississippi Valley. Eggs are laid 2 months after mating, in nest holes dug in sand located about 50 m from a water body. Clutch size varies from about 9 to 50 eggs, and success is highly variable, dependent on predation and ambient temperature. Incubation takes about 14 to 20 weeks, with hatchlings emerging in the fall. Alligator snapping turtles invest no resources on parenting. Once the eggs are laid and nests are secured, the adults return to the water. Juveniles look similar to adults, but do not reach sexual maturity until 11 to 13 years of age.

Alligator snapping turtles prefer deep, large water bodies and submerged cover (e.g., submerged logs, beaver dams, overhanging shrubs) and substantial overhead canopy. Adults prefer to remain close to these core submerged features. The average home range of alligator snapping turtles in Oklahoma is reported to be approximately 880 linear meters for females and about 480 for males. Juvenile alligator snapping turtles range further than adults, with a home range averaging about 1,000 linear meters.

3.2 Common Garter Snake

The common garter snake is one of the most abundant snakes in North America. Of the four subspecies of the common garter snake found in Texas, the Texas garter snake (*Thamnophis* *sirtalis annectens)* is the only subspecies known to inhabit eastern Texas locations; Harris County is one of several upper Gulf Coast counties in which these snakes have been observed in the last decade (Cannatella and LaDuc 2011). Regional populations of common garter snakes across the continent are distinguished mostly by variation in color patterns. The adult common garter snakes range in size between 46 and 137 cm (18 and 54 inches), and weigh an average of 150 g. The males are smaller than females and the young, which are similar in appearance to the adults, are born at 12.5 to 23 cm (5 to 9 inches) long (Zimmerman 2002).

The adaptability and resilience of the common garter snakes are evidenced by their residence in a wide variety of terrestrial and semiaquatic habitats, including meadows, marshes, woodlands, hillsides, and suburban and urban areas where debris, rock walls, foundations, gardens and other features provide good cover. These snakes prefer moist, grassy environments such as is found near the edges of ditches, ponds, lakes, and streams (Zimmerman 2002). In Texas, these snakes are found primarily in lowland habitats, particularly in areas with standing or running water, but can also be seen in open or edge habitats (Cannatella and LaDuc 2011). While these snakes tolerate a broader range of temperatures than do most, they bask in the sun during the day, and convene in coiled masses during sleep or hibernation to retain body heat. Hibernation occurs in natural cavities, rodent or crayfish burrows, under rock piles, or in stumps.

The common garter snake eats a variety of prey, dependent primarily on whether it is appropriately sized for swallowing whole. The adult diet includes amphibians, fish, and insects. Juvenile garter snakes eat a greater proportion of earthworms and insects than do adults. Baby birds, mammals, molluscs, and other snakes are also taken as prey (Cannatella and LaDuc 2011).

Garter snakes mate in the spring, as soon as they emerge from hibernation, and are ovoviviparous, meaning they carry their young until birth. In the summer and fall, the females birth an average of 26 young. The mother snakes allow the young to be around them for several days after birth, but do not provide any care, protection, or nourishment. These snakes reach sexual maturity, and maximum size, at three to four years of age, though Zimmerman (2002) indicates that the average lifespan of common garter snakes is approximately two years and that most common garter snakes probably die in their first year of life.

Common garter snakes are eaten by a wide variety of predators, including large fish, bullfrogs (*Lithobates catesbeiana*), snapping turtles, milk snakes (*Lampropeltis triangulum*), American crows (*Corvus brachyrhynchos*), hawks, great blue herons, raccoons, foxes, squirrels, and shrews. These snakes can harbor a parasitic nematode, which resides in their tails, causing a shortened tail (Zimmerman 2002).

4 BIRDS

Birds selected as receptors for the BERA include the neotropic cormorant, great blue heron, spotted sandpiper, and killdeer. These species were selected because they are expected to be present in the vicinity of the Site and represent a variety of life histories and feeding behaviors. Relevant information on each of these species is provided in the following sections. In addition, brown pelican (*Pelecanus occidentalis*), bald eagle (*Haliaeetus leucocephalus*), and white-faced ibis (*Plegadis chihi*) are discussed as they are state-listed species whose ranges include the Site vicinity.

4.1 Neotropic Cormorant

The neotropic cormorant is one of the smaller cormorants; adults weigh between 1.2 and 1.4 kg (Telfair and Morrison 2005). Neotropic cormorants are year-round residents in coastal Texas. These birds are tolerant of a range of climatic and environmental conditions and inhabit wetlands in fresh, brackish, or salt water. In coastal areas, this cormorant is associated with sheltered bays, inlets, estuaries, lagoons, and rocky outcrops. Key habitat requirements include water deep enough for diving and elevated perches in trees, shrubs, or other structures for nesting, roosting, and drying plumage after feeding. While the neotropic cormorant is capable of perching in trees or on posts, pilings, and even cables and wires, the posterior location of its short legs and its thick and laterally flattened ankles make this bird a clumsy walker on land (Telfair and Morrison 2005).

Although neotropic cormorants have been reported to occasionally consume shrimp and amphibians, their primary prey is fish smaller than 8 cm (King 1989). They appear to forage opportunistically (i.e., feeding on species that are most abundant rather than selecting specific species). Neotropic cormorants forage mainly by pursuit-diving and are the only cormorant known to plunge-dive in shallow waters (less than 2 m depth) (Telfair and Morrison 2005). Foraging area for this species is present at the Site, and the species is likely to roost nearby.

The neotropic cormorant breeds annually, producing clutches of between 1 and 4 eggs, and can reach sexual maturity by 1 year of age. There is little information available regarding home ranges for this species; post-breeding dispersal by juveniles ranges between fidelity to natal area, to dispersion several tens to hundreds of kilometers from the breeding site (Telfair and Morrison 2005). This species is tolerant of all but close and disruptive human activities.

4.2 Great Blue Heron

The great blue heron is the largest member of the heron family in North America, with body weight of males averaging slightly greater than body weight of females. A mean value of 2.2 kg for both sexes was assumed for this BERA (USEPA 1993). The great blue heron is found in freshwater and nearshore marine habitats throughout North and Central America. Habitats for great blue herons include streams, creeks, lake margins, and estuaries, with shallow water (<0.5 m) and a firm substrate on which to wade. Nearby wooded cover (within a few kilometers) is important for nesting. The great blue heron is a year-round resident of coastal Texas.

The preferred prey of great blue herons is fish; great blue herons will also eat amphibians, reptiles, crustaceans, insects, birds, and mammals (Alexander 1997; USEPA 1993). When fishing, great blue herons require shallow waters (to 0.5 m) with a firm substrate. Great blue herons consume relatively small fish that can be swallowed whole; 95 percent of fish consumed by a Wisconsin population of great blue herons were less than 25 cm in length (USEPA 1993).

In some areas, herons defend feeding territories, but in other areas, they are opportunistic and lack fidelity to a particular feeding site (USEPA 1993). Adult herons tend to feed the same type and size of food to their nestlings as they consume themselves. Predation on herons is mainly on eggs and young. Predators of young great blue heron eggs include crows and ravens. Eagles, raccoons, and hawks are among the animals which prey on the young birds and occasionally even adults (UMMZ 2011).

Great blue heron nests generally consist of a stick platform over 1 m in diameter; the nests may be reused and expanded for multiple years. Only one brood, with an average clutch size of 3 to 5 eggs, is raised per year, although if the clutch is destroyed, the parents may produce a replacement clutch. Both parents incubate and feed the young. Chicks fledge at approximately 2 months (UMMZ 2011). During the breeding season, great blue herons are monogamous and colonial. Breeding colonies are generally close to foraging grounds; a study of great blue herons in Minnesota lakes found the distance between nesting colonies and feeding sites to range from 0 to 4.2 km, averaging 1.8 km (USEPA 1993). The median flight distance to feeding areas reported in a separate study in Minnesota was 2.7 km (Custer and Galli 2002).

4.3 Spotted Sandpiper

The spotted sandpiper is the most widespread shorebird in North America (Oring et al. 1997). Adult males weigh approximately 38 g (range = 34 to 41 g), while the larger females average about 47 g (43 to 50 g) (Maxson and Oring 1980; USEPA 1993). Spotted sandpipers are relatively common winter residents in some of the local habitats around the Houston Ship Channel (Litteer 2009) and their foraging habitats are present at the Site.

The spotted sandpiper obtains much of its diet by probing or "mining" soft sediments along shorelines (USEPA 1993). This species is a generalist feeder and will occupy almost all habitats near water, including the shorelines of ponds, streams, and rivers, as well as meadows, agricultural areas, and forested areas. This species typically forages within 200 m of the shoreline (Oring et al. 1997). Spotted sandpipers are visual foragers and prey on all manner of aquatic and terrestrial invertebrates and occasionally small fish.

Spotted sandpipers are winter visitors to coastal Texas. No relevant home range or foraging range information was found for this species. An estimated foraging range of linear shoreline of 1,500 m for sanderlings, a similarly sized, invertivorous shorebird that is known to winter in coastal Texas, was used for exposure modeling, based on data for a wintering population in central California (Macwhirter et al. 2002).

4.4 Killdeer

The killdeer is a relatively large upland plover (average adult weight of 88 g, UMMZ 2011; average adult female weight of 101 g, Jackson and Jackson 2000) that feeds predominantly on terrestrial invertebrates (e.g., earthworms, beetles, grasshoppers, and other small invertebrates). The species is widespread in open areas (e.g., agricultural fields, lawns, golf courses) throughout North America and is nonmigratory across the southern United States, including Texas (Jackson and Jackson 2000). It is known to be common year-round in the vicinity of the Site (Litteer 2009).

Killdeer are primarily terrestrial invertivores. Stomach contents from killdeer in Texas were reported to contain 98 percent animal matter, mostly worms and insects (McAtee and Beal 1924).

This species is remarkably tolerant of constructed disturbances, and nesting has been documented from construction sites, road shoulders, and graveled rooftops (Jackson and Jackson 2000). Average nesting territories of killdeer in Minnesota were relatively small (0.57 acres). Larger, year-round home ranges of approximately 15 acres were reported for a northeastern California population; nesting period home ranges were smaller (Jackson and Jackson 2000). Nesting in Mississippi occurs from mid-March through late July and involves multiple broods (Jackson and Jackson 2000). The use of this surrogate species would be considered protective of smaller home range bird species at the Site (e.g., sparrows, wrens) that likely eat a larger percentage of plant matter, as well as larger omnivores (e.g., crows), and would also be protective of terrestrial ecosystem-based carnivores (e.g., hawks) that likely have larger home and forage ranges.

4.5 Brown Pelican

The brown pelican inhabits coastal areas from central North America to the Northern coasts of South America. These large seabirds, measuring from 100 to 137 cm in length and weighing approximately 2 to 5 kg, are recognized by their long bills, large gular pouch, darkly plumed body, and large (2 m) wingspans. Male pelicans are 15 to 20 percent heavier than are females and their bills are about 10 percent longer than the female's. Brown pelicans are distinct from other pelicans in that they are the only truly marine species of the pelican family.

Brown pelicans are highly social throughout the year and, in the nonbreeding season, congregate on sandbars, pilings, jetties, breakwaters, mangrove islets, and offshore rocks and islands to roost at night and rest during the day, after foraging. Breeding colonies are made up of thousands of birds, and are usually located on small, isolated estuarine, barrier or offshore islands where predation by terrestrial mammals and disturbance by humans is limited and where 30 to 50 km of a consistent foraging habitat is available (Shields 2012). In Texas, major breeding colonies are found on Pelican Island in Corpus Christi Bay and on Sundown Island in Matagorda Bay. Bird Island in Matagorda Bay, older spoil islands in West Matagorda Bay, Dressing Point Island in East Matagorda Bay, and islands in Arkansas Bay occasionally support smaller groups or colonies of breeding pelicans (TPWD 2012a). The Texas Colonial Waterbird Census (undated, as cited in Shields 2012) indicates that a brown pelican colony exists on Little Pelican Island in Galveston Bay. This species is commonly sighted, in all seasons of the year, in Upper San Jacinto Bay (Baytown Nature Center 2006). The home range of the brown pelican is limited to its foraging ground, which is generally no greater than 20 km from nesting islands. Outside of the breeding season brown pelicans in California have been observed up to 75 km from the nearest island (Shields 2012). For the purposes of the risk assessment, a home range was estimated by taking the foraging ground estimate of 20 km from the nesting island (Shields 2012) and considering that value as a radius of a circle around a nesting island that could be used for foraging, to calculate a home range area of 1,257 km^2.

Brown pelicans usually forage in shallow (<150 m) estuarine and continental shelf waters within 20 km of nesting colonies during the breeding season, and up to 75 km from nearest land during the nonbreeding season (Shields 2012). They often feed by plunging, from mid-flight, head-first into the water to retrieve prey, and primarily capture fish within the first few meters below the surface (Shields 2012). This pelican species feeds on small schooling fishes throughout its range; along the Gulf Coast menhaden and mullet are predominant in its diet (TPWD 2012a). One study along the Gulf Coast showed menhaden constituted 96 percent of the diet, with silversides, dolphinfish, and prawn contributing approximately 3, 0.8, and 0.3 percent of the brown pelican diet, respectively (Shields 2012).

The breeding season on the Texas coast lasts from March through June, with the peak breeding activities in April and May. On the Gulf Coast, nests are built on the ground, on mud banks and ledges and, in vegetation on islands covered with mangrove or other woody vegetation (TWPD 2012a). Incubation of eggs is shared between the parents; eggs are incubated under the bird's large webbed feet. Adult pelicans regurgitate predigested fish to feed hatchlings. By 3 to 4 weeks of age, the young learn to prompt adults to disgorge whole fish, which the young can swallow whole. Fledging occurs by 11 to 12 weeks of age and sexual maturity is reached by 3 to 5 years of age. Pelicans are long-lived, to approximately 30 years, and have reached 43 years of age (Shields 2012).

After decades of population declines stemming from exposures to organochlorine pesticides, the brown pelican was placed on the Federal Endangered Species list in 1970. Brown pelican reproduction subsequently improved and this species was removed from the Endangered Species List in the southeastern United States in 1985 and its population was thought to be restored along the Gulf coast by the late 1990s (Shields 2012). However, human disturbance and loss of nesting habitat continue to threaten the recovery of the brown pelican in Texas and this species still listed as "endangered" by the State of Texas (TPWD 2012a).

4.6 Bald Eagle

The bald eagle, easily recognized as an adult by its white head, dark brown plumage, and white tail, is the second largest bird of prey in North America, ranging in mass from 3.0 to 6.3 kg, with a wing span of 168 to 244 cm. Female bald eagles are 25 percent larger than are males, and both genders are smaller in the southeastern and southwestern regions of the United States than they are in northern climates (Buehler 2012). Bald eagles are opportunistic foragers, preferring to scavenge prey (carrion) or steal food from other species, but they can and will capture their own prey if these other sources of food are not available. The bald eagle prefers fish, but feeds on a variety of aquatic and terrestrial mammals, reptiles, amphibians, crustaceans, and a variety of birds, including waterfowl, gulls, and even great blue herons. A review averaging 20 studies across the bald eagle's range characterized the diet as approximately 56 percent fish, 28 percent birds, 14 percent mammals, and 2 percent other (Buehler 2012). The Texas Parks and Wildlife Department lists American coots, catfish, rough fish, and soft-shell turtles as the most common components of the bald eagle diet in Texas (TPWD 2012b).

In Texas, there are two populations of bald eagles: the breeding birds occur in the eastern half of the state and in coastal counties from Rockport to Houston; and, the wintering populations are found in the Panhandle, Central and East Texas, and in other areas where suitable habitat exists. While the majority of bald eagles observed in Texas are wintering birds that breed in northern states (TPWD 2012b), the breeding populations of bald eagles are said to be building along the Gulf Coast, in Louisiana and Texas (Buehler 2012). San Jacinto county is listed among the 47 counties in which bald eagle nests had been known to occur as of 2003 (TPWD 2012b). However, the Baytown Nature Center (2006) census data reports that this bird has been observed only in the winter months, and only rarely, in Upper San Jacinto Bay.

Information on home ranges of the bald eagle varies widely, depending on breeding status of individual, season, and most importantly, food availability. Breeding adults have been observed to occupy from 7 to 22 km^2. Non-breeding bald eagles are nomadic, relative to breeding birds, and have been observed to occupy areas ranging from 10,000 to 55,000 km^2. (Buehler 2012). Wintering eagles' ranges vary based on whether or not individuals are associated with mates. Mated pairs range within hundreds of square km, while non-mated wintering individuals might range to 4,000 km^2. An average winter range of 310 km^2 was reported for bald eagles in Colorado, with mated pairs having a smaller home range (128 km^2) than unmated eagles (average home range of 547 km^2), while in Missouri, winter home ranges of 48 and 18 km^2 were reported over two consecutive years of study (Buehler 2012).

Preferred habitats of wintering bald eagles are characterized by proximity to abundant forage associated with open water and waterfowl habitats and by availability of desirable night roost sites, such as those afforded by the oldest, tallest trees that provide unobstructed views near water, on windbreaks, and in secluded canyons (TPWD 2012b). Wintering bald eagles often congregate in large numbers in habitats that provide adequate forage and roost sites that are buffered from inclement weather and human activity (Buehler 2012).

In general, the habitat of the breeding bald eagle is the same as that of the wintering eagle. Bald eagle nests are found in forested areas, adjacent or close (<2 km) to large bodies of water and other suitable foraging habitats, and removed from human activity. Bald eagle nests are rarely found at distances <500 m from human development (Buehler 2012). While eagle nests are constructed in a variety of tree species, preferentially in the tallest tree in an area, nests in East Texas are built primarily in loblolly pine (TPWD 2012b).

The nesting season in southern latitudes (e.g., Florida and other Gulf Coast states) is somewhat prolonged over those of northern climes, ranging from late fall through early spring (Buehler 2012). In Texas, eagles nest from October to July, with peak egg-laying occurring in December and hatching occurring in January (TPWD 2012b).

4.7 White-Faced Ibis

The white-faced ibis is a medium-sized wading bird, weighing from 450 to 525 g, standing about 2 feet tall, and identified at a distance by its long neck, legs and curved bill, and uniformly dark, maroon-brown plumage. During the breeding season, the white-faced ibis is distinguished by metallic bronze, purple and green sheens to the chestnut-maroon plumage, red legs, and a reddish purple face bordered by a thin line of white feathers separating the forehead from the face and extending around the back of the eye (Ryder and Manry 1994). The white-faced ibis is listed as a threatened species by the state of Texas (TPWD 2012c).

White-faced ibis occur mainly in the western United States, breeding in marshes and irrigation areas throughout the Great Basin, most commonly in Utah, Nevada, and California. The winter range of this bird is primarily coastal Louisiana and Texas south to several Mexican states, Guatemala, and Costa Rica (Ryder and Manry 1994). It is described as a year-round resident of coastal Texas and western Louisiana (Ryder and Manry 1994), though records from the vicinity of the Site indicate that it is more of an occasional spring/summer visitor in the Site vicinity. The Baytown Nature Center, located a few miles downstream of the Site on the San Jacinto River, lists this species as rare in winter and spring, and occasional in the summer and fall seasons (Baytown Nature Center 2006).

This species primarily inhabits freshwater wetlands and marshes, as well as swamps ponds and rivers, and is commonly found feeding in flooded agricultural fields, estuarine wetlands, and temporary, shallow wetlands created by rainfall or flooding (Ryder and Manry 1994). In Texas and Louisiana, the white-faced ibis nests mostly in coastal marshes and wetlands of the outer coastal plains (Audubon 2012; Ryder and Manry 1994). The preferred roosting habitats in Texas are low platforms of dead reed stems or on mud banks (Ryder and Manry 1994; TPWD 2012c). This species has also been observed nesting on bare ground in coastal areas dominated by the shrubby coastal plant sea oxeye (*Borrichia frutescens*) (TPWD 2012c;

Ryder and Manry 1994). The white-faced ibis breeds in large colonies at established roosting sites that are used repeatedly for several years, though changing water levels will prompt colonies to switch nesting locations. In Louisiana and possibly eastern Texas, the white-faced and glossy ibis species will breed in the same colony, though they do not interbreed (Ryder and Manry 1994).

In Texas, egg-laying and incubation extend from mid-April through early July, with three or four eggs hatching after an incubation period of approximately 21 days (TPWD 2012). The parents share in incubation and brooding activities. Fledglings leave the colony after about 6 or 7 weeks, usually accompanying adults to foraging grounds (Ryder and Manry 1994).

White-faced Ibis commonly feed in large flocks of as many as 1,000 birds or more. These birds wade in areas of shallow standing water, or traverse emergent ground, foraging for aquatic and moist-soil invertebrates. Prey on the water or ground surfaces are located visually, while tactile probing with their bills is used to find prey in sediment and soils. White-faced ibis prefer shallow (5 to 15 cm) wading depths and foraging areas with emergent vegetation (Safran et al. 2000). Prey items are usually rinsed in pools of water before being eaten (Ryder and Manry 1994), although the esophagi of birds collected in Nevada contained substantial amounts of soil (Bray 1986; Bray and Klebenow 1988). A variety of prey is taken, including insects, newts, leeches, small crustaceans, worms, fish, frogs, and snails (TPWD 2012c). Stomach contents of white-faced ibis collected in Louisiana most frequently contained crayfish and insect larvae, in addition to small fish, frogs, snails, small bivalves, and earthworms (Belknap 1957).There is relatively little information available regarding the home or foraging ranges of white-faced ibis. In Nevada, during the nesting period, most birds foraged 3–6 km (but up to 18 km) from the breeding colony; while breeding adults and recently fledged young ranged 40–48 km from colonies observed in Idaho (Ryder and Manry 1994). The territory size outside of the breeding period is unknown for this species. Based on the foraging ranges described by Ryder and Manry (1994) and expert opinion, the Great Basin bird observatory suggests a recommended habitat patch size of >1,200 ha, or > 12 km^2 (GBBO 2012).

5 MAMMALS

Two semiaquatic mammals are selected as receptors for the BERA: raccoon and marsh rice rat. These species were selected because they occupy different habitats and have different feeding behaviors and life histories. In addition, Baird's pocket gopher is a proposed receptor surrogate for terrestrial mammals to be evaluated in the ecological risk assessment for the south impoundment (Appendix E). Relevant information on each of these species is provided in the following sections.

5.1 Raccoon

The raccoon is the most abundant and widespread medium-sized, omnivorous mammal in North America (USEPA 1993). Raccoons exploit a wide variety of habitats. Habitats include floodplain forests, swamps, and marshes. Raccoons are extremely adaptable to human environments and can be found in abundance in suburban residential areas and farmlands. High-quality habitat for raccoons includes sites that have access to fresh water, trees or other structures for nesting, and high food availability including fruits, grains, invertebrates, and other animals.

Adult male raccoons in an Alabama study averaged 4.3 kg and adult females averaged 3.7 kg. In a Missouri study, male raccoons averaged 6.8 kg, and females 5.7 kg. Mortality is high in young-of the year raccoons; average lifespan in the wild is 5 years, with a maximum recorded age of 16 years (UMMZ 2011).

Raccoons are highly opportunistic feeders and omnivorous, with a diet that may include carrion, garbage, birds, mammals, fish, amphibians, reptiles, grains, fruits, most food prepared for human or domestic animal consumption, agricultural crops, and invertebrates, including insects, crayfish, and mussels. Proportions of different foods in the diet depend on location and season. Plant foods dominate raccoon diets for most of the year except during spring and early summer, concurrent with the breeding season, when animal matter may be consumed more frequently (USEPA 1993). Fish ranging in size from 2 to 9 inches (5 to 23 cm) were found in the stomachs of raccoons collected in Michigan by Alexander (1977). Food ingestion rates for raccoons were not found in the literature; an allometric equation for placental mammals was used to estimate a daily ingestion rate for raccoon in the exposure model (Nagy 2001).

Raccoons escape many predators by remaining in a den during the day; they are alert and can be aggressive when active at night. Large predators may prey on raccoons, including coyotes, wolves, and owls, and their young may be taken by snakes (UMMZ 2011). Throughout most of North America, raccoons mate during February and March. Most females will produce one litter per year, and many raccoons produce litters within their first year of life. Gestation averages 63 days (Sanderson 1987), and most litters consist of three or four young. Nesting sites are primarily in hollow trees, but raccoons will also use ground dens, brush piles, and abandoned human structures for nesting, usually within a few to a few hundred meters of surface water.

Population densities are strongly dependent on habitat quality, including food availability and abundance of potential nest sites, with suburban areas generally having higher densities than rural/wild areas. Home range areas range from less than 0.05 km^2 in suburban neighborhoods to more than 5 km^2 in the wild, though values of one to a few km^2 are most commonly reported (USEPA 1993). Juvenile and adult males tend to have larger home ranges than do females (Sanderson 1987).

5.2 Marsh Rice Rat

The marsh rice rat is a semiaquatic, nocturnal, omnivorous rodent native to the southeastern U.S. This species typically inhabits marshy areas, but may be found anywhere that there is sufficient grass and groundcover to offer protection and foraging (Davis and Schmidly 1994). The marsh rice rat will readily use water to move among foraging areas and escape predation.

The marsh rice rat is considered an omnivore, with about equal parts animal and plant matter constituting its diet, though it may be more carnivorous in summer when animal prey is available (Sharp 1967). Leaves, seeds of marsh grasses and sedges, and fungus are part of the marsh rice rat's diet, and this species preys on a variety of animals including crabs, fish, insects, and bird eggs; they may occasionally scavenge carcasses of rodents and birds (Davis and Schmidly 1994). Sharp (1967) found a mixture of seeds and animal prey in stomachs of rice rats captured on a coastal island in Georgia; animal prey consisted mainly of crabs and insects including dipterans and beetle larvae.

Marsh rice rats are sexually mature at 40 to 45 days old, and weight from about 40 to 68 g as adults (average weight of 51 g; Davis and Schmidly 1994; average adult female weight of 67.70 g ± 0.85 g [±standard error], Fernandes 2011). They can reproduce year round, and a female may produce five or six broods per year, consisting of two to seven offspring. Home ranges of 0.37 hectares for males and 0.23 hectares for females have been reported; average range lengths include 75 m for a Maryland population and 68 and 82 m for marsh rice rats in Florida (Wolfe 1982).

5.3 Baird's Pocket Gopher

The Baird's pocket gopher, also known as the Louisiana pocket gopher, is virtually indistinguishable, morphologically, from the plains (*G. busarius*) and Attwater's (*G. attwateri)* pocket gophers, each of which inhabit different regions of Texas (Sulentich et al. 1991; TPWD 2011c). These pocket gophers are small, dark brown, burrowing herbivores. With long, curved, and specially adapted front claws, a broad, flat head, tiny, bead-like eyes and rudimentary ears, and a compact body with skin and hair arranged to allow movement through borrows both backward and forward, these gophers are more highly specialized for digging than any other North American rodent (TPWD 2011c; KSR 2011; Sulentich et al. 1991). *G. breviceps* is the smallest of its congenerics, averaging 208 mm in length and weighing between 78 and 150 g, with an average reported weight of 100 g (MNH 2012). The Baird's pocket gopher is found in the eastern portion of Texas and has been found on both sides of the San Jacinto River in Harris County (Sulentich et al. 1991; TWPD 2011c).

Geomys live underground most of their lives and maintain labyrinths of burrows in sandy and loamy soils, digging to an average depth of approximately 6 inches and up to 2 feet, generally on treeless land (TPWD 2011c). As much of the burrowing is done in search of food, tunnels meander through feeding areas, and can extend well over 100 m. These rodents are solitary; each tunnel system is occupied by only one gopher. They rarely leave their burrows, except at night for mating or for limited foraging beyond the entrance (KSR 2011). In wet months, pocket gophers are known to live and nest in above-ground mounds of dirt, in order to avoid being flooded out of their burrows and tunnels (Sulentich et al. 1991).

The Baird's pocket gopher is herbivorous, obtaining most of its food while digging tunnels and feeding primarily on underground roots and the stems of weeds and grasses. While most plant food is encountered and ingested while the gopher digs its lateral tunnels, green plants and grasses are obtained at night from around the entrance of the tunnels and beyond. Fur-lined cheek pouches are used to carry food and nesting material. Cellulose-digesting bacteria in the digestive system help the Baird's pocket gopher digest grasses and stored underground rhizomes during the winter and these gophers, as do many rodents, increase their utilization of food by re-ingesting their fecal pellets (Sulentich et al. 1991; TPWD 2011c).

The Baird's pocket gopher begins breeding in eastern Texas in early February and continues through August, with peak productivity occurring in June and July. One to four young are born to each litter (Sulentich et al. 1991). As with most rodents, the newborns are nearly naked, with eyes and ears closed, and are helpless at birth. The young remain with their mother until nearly full-grown, at about 6 to 7 weeks of age, when they disperse to lead an independent life (TPWD 2011c). Sexual maturity is reached within 90 days of birth (Sulentich et al. 1991).

In east Texas, Baird's pocket gophers are preyed on by long-tailed weasels, and, when caught out of their burrows, are vulnerable to king snakes (*Lampropeltis getula*), great-horned owls (*Bubo virginianus*), red-tailed hawks (*Buteo jamaicensis*), and striped skunks (*Mephitis mephitis*), among other common rodent predators (Sulentich et al. 1991; TPWD 2011). Because they remain protected in their burrows most of the time, pocket gophers are long-lived relative to many other rodents, living an average of 1 to 2 years in the wild (TPWD 2011c). The estimated population density in prairie habitat near College Station, Texas, was approximately 0.55 gophers per hectare (Sulentich et al. 1991).

5.4 Virginia Opossum

The Virginia opossum (*Didelphis virginiana)* is a widespread and adaptable nocturnal scavenger similar in size to a large house cat. (UMMZ 2003). It is the only marsupial found north of Mexico. Opossums range from Central America through much of the continental United States, including the eastern two-thirds of the country and the coastal Pacific. Opossums range in size from 350 to 940 mm, averaging 740 mm. Adult males weigh an average of 5.5 pounds, and adult females average 4.0 pounds (Georgia DNR 2012); size may vary with location and climate (MNH 2012). The lifespan of a Virginia opossum averages 2 years, though many die in the first year of life (TPWD 2012d). Both northern and southern populations have white fur with black tips. They have a pointed snout, opposable thumb-like appendages and a scaly prehensile tail that can be used to climb, hang, or grasp objects (TPWD 2012c). The opossum is known for its tendency to "play dead": when exposed to a threatening situation, the opossum can enter a catatonic state in which its breathing nearly stops. The behavior is considered to be an involuntary defense mechanism (MNH 2012).

Opossums are well adapted to living near humans and occur in a variety of habitat types. They are primarily found in woodland areas especially near creeks, rivers, or lakes, but can also occupy marshes, farmland, prairies, and urban and rural environments. They prefer to live in hollow trees and logs, but can also nest under rocks, buildings, bridges, attics, woodpiles, or in other animals' abandoned burrows (UMMZ 2003; Georgia DNR 2012). In east Texas, Virginia opossums typically frequent overlapping home ranges approximately 0.05 km^2 in size, although the minimum size of home ranges may vary from 0.001 to 0.23 km^2. In East Texas woodland habitat, the density of opossums is about one opossum every 0.02 km^2 while in sandy, coastal parts of the state the density is about one opossum every 0.06 km^2 (Davis and Schmidly 1994).

The Virginia opossum has a brief gestation period of 2 weeks, after which the relatively undeveloped young crawl from the birth canal and attach themselves to the mother's nipple inside of her fur-lined pouch, where they stay attached for 7 weeks of nursing (UMMZ 2003). Litters usually consist of seven young, and Virginia opossums typically have two litters per year (Georgia DNR 2012).

Virginia opossums are omnivorous. Consuming mostly insects and carrion, the opossum also forages for acorns, berries, and other fruit and is also known to eat crustaceans, frogs, bird eggs and nestlings, small rodents, and the young of its own kind. In human-populated areas, the opossum is known to scavenge for garbage and can be considered a nuisance for this reason (Georgia DNR 2012).

Common predators of Virginia opossums include canids, raccoons, and raptors. Humans are also a main cause of mortality through hunting and trapping, and opossums are frequently killed on roads (Georgia DNR 2012). Opossums are considered a game animal and in many states there are rules and regulations pertaining to their harvest through trapping and hunting. Despite their appeal to hunters, biologists do not believe that hunting is a threat to most populations of this species (Georgia DNR 2012).

6 REFERENCES

Alexander, G.R., 1997. Food of vertebrate predators on trout waters in north central lower Michigan. *Mich. Academ.* 10:181–195.

Baytown Nature Center, 2006. BNC Bird Checklist. Compiled by D. Litteer, 2000 – 2006. Accessed on February 3, 2012 at: http://www.baytownnaturecenter.org/

Belknap, H. W., 1957. Observations on the white-faced ibis (*Plegadis chihi*) in Louisiana. Master's Thesis. Louisiana State University, Baton Rouge. Cited in Ryder and Manry 1994.

Bray, M.P., 1986. Feeding ecology of white-faced ibises in the Lahontan Valley, Nevada. P-R Rep., Federal Aid Proj. W-53-R, Study I, Jobs 02. Nevada Dept. Wildl., Reno. Cited in Ryder and Manry 1994.

Bray, M.P., and D. A. Klebenow, 1988. Feeding ecology of white-faced ibises in a Great Basin valley, USA. Colon. *Waterbirds* 11:24-31. Cited in Ryder and Manry 1994.

Buehler, D.A., 2012. Bald Eagle. The Birds of North America Online (A. Poole, Ed.) Ithaca: Cornell Laboratory of Ornithology; Retrieved February 3, 2012, from The Birds of North America Online database: http://bna.birds.cornell.edu/bna/species/506/articles/introduction

Cannatella, D., and T. LaDuc, 2011. Herps of Texas, Texas Natural Science Center, University of Texas, Austin. On-line publication accessed at: http://www.herpsoftexas.org/content/common-gartersnake on October, 24, 2011.

Custer, C.M., and J. Galli, 2002. Feeding habitat selection by great blue herons and great egrets nesting in east central Minnesota. *Waterbirds* 245:115-124.

Davis, W.B., and D.J. Schmidly, 1994. The Mammals of Texas: Online Edition. Accessed at http://www.nsrl.ttu.edu/tmot1/Default.htm. Accessed on September 27, 2011.

Fernandes, M.V., 2011. Effects of changes in the Everglades on two indicator species: *Sigmodon hispidus* and *Oryzomys palustris*. Open Access Dissertations. Paper 606.

Franklin, C., 2011. www.Texasturtles.org, Texas Turtle Information. Accessed at: http://www.texasturtles.org/Macrochelys_temminckii.html. Accessed on October 11, 2011.

Froese, R. and D. Pauly, 2009. FishBase: World Wide Web electronic publication. www.fishbase.org, version (07/2009). http://www.fishbase.org. Accessed on October 10, 2011.

Fuller, P., and L.A. Somma, 2011. *Macrochelys temminckii*. USGS Nonindigenous Aquatic Species Database, Gainesville, FL. http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=1227. Revision Date: 10/28/2009. Accessed on October 11, 2011.

FWRI, 2010. Black Drum *(Pogonias cromis)* Florida Fish and Wildl. Conserv. Comm., accessed at: http://myfwc.com/media/194648/black_drum.pdf on October 10, 2011.

GBBO, 2012. White-faced ibis, *Plegadis chihi*, conservation profile. http://www.gbbo.org/pdf/bcp/20_White-faced%20Ibis.pdf. Accessed July 16, 2012. Great Basin Bird Observatory.

Georgia DNR, 2012. Georgia Department of Natural Resources, Wildlife Services Division. Accessed at http://www.georgiawildlife.com/node/937 on July 30, 2012.

Hassan-Williams, C., T.H. Bonner, and C. Thomas, 2011. Texas Freshwater Fishes. Texas State University – San Marcos, Department of Biology. Accessed at http://www.bio.txstate.edu/~tbonner/txfishes/index.htm, on October 6, 2011.

Hill, K, 2005. Smithsonian Marine Station at Fort Pierce. Animals of the Indian River Lagoon: *Parolythes lethostigma.* http://www.sms.si.edu. Accessed October 10, 2011.

Jackson, B.J., and J.A. Jackson, 2000. Killdeer *(Charadrius vociferus)*, The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/517.

Jackson, C.G., and A. Ross. 1971. The occurrence of barnacles on the alligator snapping turtle (*Macroclemys temminckii*). *Journal of Herpetology* 5:188-189.

King, K.A., 1989. Food habits and organochlorine contaminants in the diet of olivaceous cormorants in Galveston Bay, Texas. Southwest. *Nat.* 34:338-343.

KSR, 2011. Mammals of Kansas, Plains pocket gopher (*Geomys bursarius*) Online publication accessed at: http://www.ksr.ku.edu/libres/mammals_of_kansas/geomys-burs.html. Accessed on October 11, 2011.

Litteer, D., 2009. Baytown Nature Center Bird Checklist. http://www.baytown naturecenter.org/publications_information/bird_checklist.html. Accessed January 7, 2009.

Macwhirter, B., P. Austin-Smith, Jr., and D. Kroodsma, 2002. Sanderling (*Calidris alba*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/653. Accessed October 26, 2011.

Maxson, S.J., and L.W. Oring, 1980. Breeding season time and energy budgets of the polyandrous spotted sandpiper. *Behaviour* 74:200–263.

McAtee, W.L., and F.E.L. Beal, 1924. Some common game, aquatic, and rapacious birds in relation to man. *U.S.D.A. Farmers' Bull.* No. 497(revised) (not seen, as cited in Jackson and Jackson 2000)

MNH, 2012. North American Mammals, Online publication, Costello, R. and A. Rosenberger, Eds., Smithsonian National Museum of Natural History. Accessed at: http://www.mnh.si.edu/mna/image_info.cfm?species_id=65 on July 30, 2012. Museum of Natural History.

National Geographic, 2011. Alligator Snapping Turtle. Online publication accessed at http://animals.nationalgeographic.com/animals/reptiles/alligator-snapping-turtle/ on October 11, 2011.

Nichols, M., J. Pruitt, D. Munsey, G. Good, B. Meyer, K. Urban, K. Francl, and P. DiLaura, 1999. *Macrochelys temminckii* (On-line), Animal Diversity Web. Accessed October 11, 2011. http://animaldiversity.ummz.umich.edu/site/accounts/information/Macrochelys_temminckii.html.

Oring, L.W., E.M. Gray, and J.M. Reed, 1997. Spotted Sandpiper *(Actitis macularius)*, The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/289

Purdue, J.R., and H. Haines, 1977. Salt water tolerance and water turnover in the snowy plover. *Auk* 94:248-255.

Rozas, L.P., and R.J. Zimmerman, 2000. Small-scale patterns of nekton use among marsh and adjacent shallow nonvegetated areas of Galveston Bay Estuary, Texas (USA). *Marine Ecology Progress Series* 193:217-239. (not seen, as cited in Hassan-Williams et al., 2011)

Ryder, R.A., and D.E. Manry, 1994. White-Faced Ibis (*Plegadis chihi*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology; Retrieved July 10, 2012 from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/130

Safran, R.J., M.A. Colwell, C.R. Isola, and O.E. Taft, 2000. Foraging site selection by nonbreeding white-faced ibis. *The Condor* 102:211-215.

Sanderson, G.C., 1987. Raccoon. In: *Wild furbearer management and conservation*. M.Novak, J.A.Baker, M.E.Obbarel (eds). M. Novak, J.A. Baker, M.E. Obbarel, et al. (eds). University of Pittsburgh Press, Pittsburgh, PA. pp. 487–499.

SCDNR, 2011. ACE Basin Species Gallery, South Carolina Department of Natural Resources. Online publication accessed at: http://www.dnr.sc.gov/marine/mrri/acechar/specgal/flounder.htm. Accessed October 10, 2011.

Schofield, P.J., and P. Fuller, 2011. *Fundulus grandis*. USGS Nonindigenous Aquatic Species Database, Gainesville, FL. Accessed at http://nas.er.usgs.gov/queries/factsheet.aspx?SpeciesID=687, Revision Date: 2/8/2006, on October 6, 2011.

Sharp, H.F., 1967. Food ecology of the rice rat, *Oryzomys palustris* (Harlan) in a Georgia salt marsh. *J. Mammal*. 48: 557-563.

Shields, M., 2012. Brown Pelican (*Pelecanus occidentalis*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of Ornithology. Available at: http://bna.birds.cornell.edu/bna/species/609/articles/behavior?searchterm=Shields

Sulentich, J.M., L.R. Williams, and G.N. Cameron, 1991. Mammalian Species No 383, *Geomys breviceps*. The American Society of Mammalogists, November 6. Accessed at http://www.science.smith.edu/departments/Biology/VHAYSSEN/msi/pdf/i0076-3519-383-01-0001.pdf, on October 20, 2011.

Telfair II, R.C., and M.L. Morrison, 2005. Neotropic Cormorant (*Phalacrocorax brasilianus*), The Birds of North America Online (A. Poole, Ed.). Ithaca: Cornell Lab of

Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/137

TPWD, 2011a. Black Drum (*Pogonias cromis)* Texas Parks and Wildlife, Electronic publication accessed at http://www.tpwd.state.tx.us/huntwild/wild/species/blackdrum/. Accessed on October 10, 2011.

TPWD, 2011b. Southern Flounder (*Paralichthys lethostigma).* Texas Parks and Wildlife, Electronic publication accessed at http://www.tpwd.state.tx.us/huntwild/wild/species/flounder/. Accessed October 10, 2011

TPWD, 2011c. The Mammals of Texas – Online Edition, Texas Parks and Wildlife Department accessed at: http://www.nsrl.ttu.edu/tmot1/geomburs.htm, on October 11, 2011.

TPWD, 2012a. Eastern Brown Pelican (*Pelecanus occidentalis occidentalis).* Texas Parks and Wildlife, Electronic publication accessed on February 3, 2012 at: http://www.tpwd.state.tx.us/publications/pwdpubs/media/pwd_bk_w7000_0013_eastern_brown_pelican.pdf

TPWD, 2012b. Bald Eagle (*Haliaeetus leucocephalus).* Texas Parks and Wildlife, Electronic publication accessed at: http://www.tpwd.state.tx.us/publications/pwdpubs/media/pwd_bk_w7000_0013_bald_eagle.pdf. Accessed on February 6, 2012.

TPWD, 2012c. White-Faced Ibis (*Plegadis chihi*). Texas Parks and Wildlife, Electronic publication accessed at: http://www.tpwd.state.tx.us/huntwild/wild/species/ibis/. Accessed on July 10, 2012.

TPWD, 2012d. Virginia Opossum (*Didelphis virginiana*). Texas Parks and Wildlife, Electronic publication accessed at: http://www.tpwd.state.tx.us/huntwild/wild/species/oposum/ Accessed on July 30, 2012.

UMMZ, 2003. *Didelphis virginiana*, Animal Diversity Web. Accessed at http://animaldiversity.ummz.umich.edu/site/accounts/information/Didelphis_virginiana.html on July 20, 2012.

UMMZ, 2011. Diversity Web. http://animaldiversity.ummz.umich.edu. Accessed September 26, 2011. University of Michigan, Museum of Zoology, Ann Arbor, MI.

USEPA, 1993. Wildlife exposure factors Handbook. Volume I of II. EPA/600/R-93/187a. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

Wolfe, J.L., 1982. *Oryzomys palustris*. Mammalian Species No. 176, pp. 1-5. *American Society of Mammalogists*. 18 June 1982.

Zimmerman, R., 2002. *Thamnophis sirtalis* (On-line), Animal Diversity Web. Accessed October 24, 2011, at: http://animaldiversity.ummz.umich.edu/site/accounts/information/Thamnophis_sirtalis.html.

APPENDIX B
ECOTOXICITY PROFILES

APPENDIX B
ECOTOXICITY PROFILES
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
AhR	aryl hydrocarbon receptor
AVS	acid-volatile sulfide
AWQC	ambient water quality criteria
BEHP	bis(2-ethylhexyl)phthalate
BERA	baseline ecological risk assessment
bw	body weight
CCC	Criterion Continuous Concentration
CMC	Criterion Maximum Concentration
$COPC_E$	chemical of potential ecological concern
CTR	critical tissue residue
dw	dry weight
EC_{50}	median effective concentration
EcoSSL	ecological soil screening level
ER-L	effects range-low
ER-M	effects range-median
EROD	ethoxyresorufin-*O*-deethylase
LC_{50}	median lethal concentration
LD_{50}	median lethal dose
LOAEC	lowest-observed-adverse-effect concentration
LOAEL	lowest-observed-adverse-effect level
LOEC	lowest-observed effect concentration
NOAEC	no-observed-adverse-effect concentration
NOAEL	no-observed-adverse-effect level
NOEC	no-observed-effect concentration
PCB	polychlorinated biphenyl
PCDD	polychlorinated dibenzo-*p*-dioxin
PCDF	polychlorinated dibenzofuran
PeCDF	pentachlorodibenzofuran

RATL	Canadian Wildlife Service's Database of Reptile and Amphibian Toxicology Literature
SJRWP	San Jacinto River Waste Pits
SSD	species sensitivity distribution
TCDD	tetrachlorodibenzo-*p*-dioxin
TCDF	tetrachlorodibenzofuran
TCEQ	Texas Commission on Environmental Quality
TEF	toxic equivalent factor
TEQ	toxic equivalent
TEQ_P	toxic equivalent for PCBs
TRV	toxicity reference value
USEPA	U.S. Environmental Protection Agency
ww	wet weight

1 INTRODUCTION

This appendix provides summary ecotoxicity information for the chemicals of potential ecological concern (COPCES) identified for the San Jacinto River Waste Pits (SJRWP) Site baseline ecological risk assessment (BERA). The profiles presented in this appendix provide the ecological effects evaluation performed for this BERA consistent with U.S. Environmental Protection Agency (USEPA) guidance (USEPA 1997). Each profile briefly describes the potential toxicity of each chemical or group of chemicals, addressing only those receptors for which the chemical is considered a COPCE. Each profile describes the types of potential toxic effects associated with exposure of terrestrial and aquatic biota to these chemicals, and provides the sources and supporting rationale for selection of individual toxicity reference values (TRVs). This section outlines the specifications for the scope of these toxicity profiles and methods for obtaining TRVs; subsequent sections provide the toxicity information needed for the BERA.

1.1 Receptor Surrogates

TRVs are identified for species considered representative of site-specific receptors or receptor surrogates. Site-specific receptor surrogates were selected to represent the following:

- Benthic macroinvertebrate communities
- Fish
  - Gulf killifish (*Fundulus grandis*)
  - Black drum (*Pogonias cromis*)
  - Southern flounder (*Paralichthys lethostigma*)
- Reptiles
  - Alligator snapping turtle (*Macrochelys temminckii*)
- Piscivorous birds
  - Neotropic cormorant (*Phalacrocorax brasilianus*)
  - Great blue heron (*Ardea herodias*)

- Invertivorous birds
  - Spotted sandpiper (*Actitis macularius*)
  - Killdeer (*Charadrius vociferous*)
- Semiaquatic mammals
  - Raccoon (*Procyon lotor*)
  - Marsh rice rat (*Oryzomys palustris*).

TRVs to address risks to these receptor groups are required for the BERA. Table B-1 provides a summary of the COPC$_E$s for each ecological receptor group. Sections below address only those receptor–COPC$_E$ pairs shown in Table B-1.

1.2 Measures of Effect

Measurement endpoints and risk questions for the BERA are outlined in the RI/FS Work Plan (Anchor QEA and Integral 2010), and are discussed in the main text of the BERA. In summary, the following types of TRVs are needed for the BERA:

- Benthic macroinvertebrates
  - Bulk sediment concentration (mg/kg) for the benthic macroinvertebrate community
  - Concentrations in water (mg/L)
  - Critical tissue residue (CTR) values for dioxin and furan compounds (or other organics) expressed as concentration in whole clams (mg/kg wet weight [ww] or lipid)
- Fish
  - Concentrations in water (mg/L)
  - CTR values for dioxin and furan (or other organics) compounds expressed as concentrations in whole fish (mg/kg ww or lipid)
  - Concentrations of metals in media ingested by fish (mg/kg dry weight [dw])
- Reptiles, birds, and mammals
  - Daily ingested doses (mg/kg-day) for reptiles and mammals for all COPC$_E$s, and for birds for COPC$_E$s other than dioxins and furans

- CTR values for 2,3,7,8-tetrachlorinated dibenzo-*p*-dioxin toxic equivalent (TEQ) concentrations in bird eggs (mg/kg ww).

As noted in the RI/FS Work Plan, the toxicity literature is often limited to studies reporting effects at the individual level. For this BERA, the types of individual effects measures are those clearly relating to population-level responses. These are generally survival, growth, and reproduction of tested individuals. Studies documenting an effect of a toxicant on an endpoint that is related by the authors of the study to survival, growth, or reproduction are also used (e.g., a developmental endpoint that is clearly related to the reduced survival of young). Studies addressing unrelated endpoints (e.g., cellular or biochemical alterations or gene expression) are generally not used to establish TRVs for the BERA, because these effects cannot be related to population-level assessment endpoints.

TRVs may be expressed as ingested doses, CTRs, concentrations in foods (fish only), or water concentrations, depending on the endpoint and receptor being evaluated. To calculate TRVs as an ingested dose, (i.e., mg/kg body weight [bw] per day) and where original toxicity studies report effect levels as concentrations in food of test animals but do not report body weight and/or consumption rate, values used by Sample et al. (1996) for body weight and consumption rate provided the basis for conversions to dose values, where needed. Table B-2 lists test species on which TRVs for birds and mammals were based and the default values for body weight and consumption rate for each. Dietary concentrations are presented in this appendix as dry weight, unless otherwise noted.

1.3 The Use of Uncertainty Factors in Estimates of Effects

The preferred approach for selecting TRVs is to find values that meet acceptability criteria (Section 1.4) and are taxonomically relevant and appropriate to the receptors of concern, but data may not be available for a given taxon or effect level of interest (e.g., a median lethal concentration [LC_{50}] may be available, but not a no-observed-adverse-effect level [NOAEL] or lowest-observed-adverse-effect [LOAEL]). In these cases, the application of an uncertainty factor to conservatively estimate the benchmark or TRV may be considered. In a review of the types and uses of uncertainty factors, Chapman et al. (1999) conclude that an uncertainty factor should account for the uncertainty in the extrapolation, but should not be so large that it renders the resultant value meaningless for assessing risk.

Chapman et al.'s (1999) review emphasizes the importance of evaluating the substance and context of the uncertainty. They caution against the extrapolation of LOAELs to NOAELs because there can be substantial uncertainty in moving from effects to no-effects concentrations. They provide several examples that support the use of uncertainty factors of 10 or less for individual extrapolations, including extrapolation of acute lethality toxicity tests to thresholds for sublethal effects in aquatic systems, and lowest-observed-effect concentration (LOEC) to no-observed effect concentration (NOEC) ratios for wildlife criteria (Chapman et al. 1999). This review points out that uncertainty factors are essentially screening tools for which the imprecision cannot be quantified, and should not be regarded as mathematical absolutes. These recommendations were used as a basis for the application of uncertainty factors in deriving TRVs where relevant effects level values were missing but related values were available.

1.4 Information Search Methods

These toxicity profiles draw from several well-established sources of TRVs and toxicity information commonly used at Superfund sites. Searches in the primary literature were also used to find information less readily available, to obtain the most recent information, and for chemicals for which it is necessary to evaluate toxicity in greater depth (e.g., dioxins and furans). The following describes the resources used to locate TRVs for the BERA.

1.4.1 Primary Literature and Compendia of Information for Superfund Sites

For bird and mammal TRVs for metals, this BERA draws largely from two widely accepted reviews of TRVs:

- The ecological soil screening levels (EcoSSLs) developed by USEPA (2005a)
- Sample et al. (1996).

Literature cited by USEPA in support of the EcoSSL values received careful and systematic scrutiny using specific and widely-used data quality criteria and with oversight by a large panel of scientists (USEPA 2005g). Therefore, documents supporting the EcoSSL values were consulted as the preferred source for identifying the bird and mammal TRVs for metals.

Derivation of TRVs for aquatic life relied primarily on the following three references:

- Draft final BERA for Portland Harbor (Windward 2011a) (fish only)
- Long et al. (1995) (summarized in NOAA 1999; benthic macroinvertebrates only)
- USEPA ambient water quality criteria (AWQC), which are considered protective of 85 percent of aquatic species (Table B-3).

The draft final BERA for Portland Harbor in Portland, Oregon (Windward 2011a) presents a detailed review of the aquatic toxicological literature for many of the $COPC_{ES}$ selected for fish at SJRWP. Their reviews include analysis of all published studies that could reasonably be found through literature searches, and selection of one or more values from the published data. The Portland Harbor BERA is in review by USEPA (draft final) and has had substantial USEPA input in its development. This compendium of information was used in this risk assessment to develop TRVs expressed as concentrations in fish foods. When well-established TRVs from Superfund sites, Sample et al. (1996), or the EcoSSL datasets are identified for use in this BERA, Integral did not conduct an independent evaluation of data quality using the original literature.

For benthic macroinvertebrates, a compilation of marine sediment benchmarks by Long et al. (1995) was used. Although other sources of marine sediment quality guidelines are available (MacDonald et al. 1996) and may be more robust on the basis of the methods used for their derivation, Long et al. (1995) is the same source of information used by the Texas Commission on Environmental Quality (TCEQ) in establishing sediment screening benchmarks for benthos. TCEQ interprets sediment chemistry in terms of risk to benthic invertebrate communities relative to Long et al.'s (1995) sediment benchmarks as follows:

- The effects range-low (ER-L) values are concentrations below which adverse effects on benthic communities rarely occur
- The effects range-median (ER-M) values are concentrations above which adverse effects on benthic macroinvertebrate communities are "probable"

- At concentrations between the ER-L and ER-M, adverse effects on benthic invertebrates are considered possible.

Although Long et al.'s (1995) ER-L and ER-M values have technical flaws (e.g., Sampson et al. 1996a, 1996b; Becker and Ginn 2008), they are regarded by TCEQ as protective of benthic communities. Therefore, in this risk assessment and consistent with the role of SQGs as screening benchmarks, ER-Ls were used to identify $COPC_{ES}$ and stations posing negligible risk to benthic macroinvertebrate communities. When concentrations of a $COPC_{E}$ in sediment exceed its respective ER-M value, the number of exceedances and area involved are considered to determine whether additional toxicity information is warranted to better describe risk.

When ER-L/ER-M values or other TRVs expressed as a bulk sediment concentration were not available for benthic macroinvertebrates, USEPA's AWQC for protection of aquatic life were used. AWQC are concentrations in water protective of 95 percent of species in an aquatic community, and are expressed as concentrations in water, as follows:

- The Criterion Maximum Concentration (CMC) is expected to be protective of aquatic life if the 1-hour average concentration in the waterbody does not exceed the CMC more than once every 3 years.
- The Criterion Continuous Concentration (CCC) is expected to be protective of aquatic life if the 4-day average concentration in the waterbody does not exceed the CMC more than once every 3 years.

If TRVs as sediment concentrations were not available to evaluate risk to benthic macroinvertebrate communities, a concentration in pore water was estimated using equilibrium partitioning models and was compared to the CCC from the available AWQC.

When AWQC and ER-L/ER-M values were not available to perform a screening comparison for fish and invertebrates, the following sources were consulted in descending order of preference:

- USEPA's ECOTOX database[1] (selected values were only from tests with marine species)
- Literature search for sediment or water toxicity data using the same databases listed above for toxicity data for birds and mammals.

When TRVs were not available from the sources listed above, a literature search for NOAEL and LOAEL values related to survival (or conversely mortality), growth, or reproductive endpoints was conducted using the following databases: Toxline, BIOSIS, Academic Search Complete, AGRICOLA, and GreenFile.

When a literature review was conducted, abstracts were reviewed to determine if an article reported on survival, growth, or reproductive endpoints for the relevant taxonomic group. Articles addressing these endpoints were retrieved for review and were evaluated according to the acceptability criteria described later in this section.

1.4.2 Toxicity Literature for Reptiles

The resources described above were consulted for information on the toxicity of $COPC_{ES}$ to reptiles. The availability of toxicity literature on reptiles is generally poor. One database of toxicity information has been developed specifically to house information on toxicity of chemicals to reptiles: The Canadian Wildlife Service's Database of Reptile and Amphibian Toxicology Literature (RATL) (Pauli et al. 2000). RATL is an annotated bibliography of literature related in some way to the exposure of reptiles to or toxicity of various chemicals, mostly metals, to reptiles and amphibians. It was searched for studies on reptiles that met the following criteria: laboratory study with exposure defined as environmental, dermal, oral, or injections, for all endpoints. Because RATL is now more than 10 years old, additional literature searches on the toxicity of $COPC_{ES}$ in reptiles were conducted.

There is a small body of literature that evaluates turtle tissue as a biomarker for organochlorine and metal contaminants (Beresford et al. 1981; Bergeron et al. 1994; de Solla et al. 1998; Keller et al. 2004; Robinson and Wells 1975). Most of the literature describes concentrations of chemicals in field-collected organisms but provides no means of

[1] http://www.epa.gov/med/Prods_Pubs/ecotox.htm

interpreting such information. The methods used to administer toxicants to turtles in most studies, such as painting of toxicants on to the surface of eggs, are not environmentally realistic, or have resulted in unrealistically high concentrations, and can be difficult to interpret for the purposes of risk assessment.

Very few toxicity studies addressing the $COPC_{ES}$ were located for reptiles in general, and none were found that presented daily ingested doses for comparison to exposure estimates for reptiles using the Site. None of the data found met the acceptability criteria for toxicity studies to be used in determining TRVs (below). The paucity of toxicity data for reptiles that is useful for risk assessment is confirmed by Weir et al. (2010) and Sparling et al. (2000a). These authors performed general literature reviews for ecotoxicological literature from 1978 to 1998 on vertebrates, and found that, of over 12,000 titles, only 163 papers address reptiles. The subject of most of these was turtles. About one-quarter of the studies address metals, a quarter address pesticides, 20 percent address polychlorinated biphenyls (PCBs), and the rest address "other" chemicals and effects of plastic and other debris.

Ecotoxicology of Amphibians and Reptiles (Sparling et al. 2000b) includes chapters on metals (Linder and Grillitsch 2000) and organic chemicals other than pesticides (Portelli and Bishop 2000). For both categories, these investigators found that the majority of information is from uncontrolled studies using field-collected organisms in which exposure to several xenobiotics could have occurred or are documented to have occurred. Most studies report tissue concentrations, not dose-response information. According to Linder and Grillitsch (2000, p. 398), "[t]here is a collective agreement that for reptiles little to no explicit information on the toxicological effect potential is available for any metal." Similarly, the majority of publications report dioxin, furan, and PCB concentrations in tissues of reptiles collected in the field. Bis(2-ethylhexyl)phthalate (BEHP) is not mentioned in the book (Sparling et al. 2000b).

In their review, Portelli and Bishop (2000) found no reports of reptiles dying as a result of PCB, dioxin, or furan exposure, despite fairly elevated concentrations in their tissues. Bishop et al. (1991) reported developmental abnormalities (e.g., abnormal eyes, claws, and bills) and behavioral abnormalities in turtles exposed to dioxins, furans, PCBs, and organochlorine pesticides, but dose-response relationships have not been reported. This and other studies cited by Portelli and Bishop (2000) suggest correlations between concentrations of PCBs, polychlorinated dibenzo-*p*-dioxins (PCDDs), and polychlorinated dibenzofurans (PCDFs) and abnormalities in developing embryos, but these data are confounded by the presence of pesticides and other chemicals in the environment and tissues of organisms studied. Portelli and Bishop (2000) note that there is no correlation between dioxin, furan, and PCBs in eggs and incidence of abnormalities when TEQ was used to characterize exposure, regardless of the toxic equivalent factor (TEF) scheme used. The available information on toxicity of dioxins and dioxin-like compounds to reptiles is reviewed in Section 2.

Because of the lack of information on the toxicity of $COPC_{ES}$ to reptiles, TRVs were not developed for reptiles, and risks to reptiles at the Site cannot be evaluated using available methods and data. Exposure assessment is conducted, and risks to reptiles are discussed in the uncertainty analysis.

1.5 Acceptability Criteria

The toxicity literature reflects a wide range of investigator objectives, most of which were not associated with ecological risk assessment. As a result, the technical quality of toxicological studies potentially available for risk assessment varies widely. Some of the available literature is not acceptable for use in a BERA. Because most studies, especially older research, provide imperfect ecotoxicological information, guidelines to evaluate the acceptability of literature used to derive TRVs are needed. The use of basic standards for data quality ensures that the meaning and uses of the reported information are clear. The following are among the most important considerations for inclusion of toxicity data in the BERA:

- Methods must be clearly presented and complete.
- The test subjects should not have been exposed to toxicants other than the toxicant under study prior to or during the investigation, unless the pre-existing exposure is addressed by the study. For field studies in which test subjects have been exposed to other chemicals, NOAELs can be derived.
- Exposures to toxicants in water are not performed with solutions within which the toxicant concentration exceeds its water solubility.
- Either an effects level (e.g., LOAEL) or a no-effects level (e.g., NOAEL) is reported.

- Investigators use and report results for an experimental control, and control media are identical to exposure media in every way except for the toxicant under study.
- The statistical design employs an appropriate number of replicates, treatments are randomized, and the level of significance is reported for differences in response from controls.
- The tested endpoint is clearly related to the survival, growth, or reproduction of the tested subjects.
- There are no obvious confounding factors, such as limited feeding of tested specimens, which could affect the test endpoint.

To the extent that a study that is selected to support the risk evaluation deviates from these guidelines, the uncertainties associated with the TRV, and therefore the risk evaluation, tend to increase.

In addition to the above guidelines, preference is given to toxicity studies with the following characteristics:

- Both a LOAEL and a NOAEL are reported.
- The form of the test chemical is reported, and is a form commonly found in the environment.
- Tissue residue-based TRVs report concentrations for whole-body samples (because concentrations of individual organs and isolated tissues such as liver or gill tissue of receptors at the Site cannot be reliably predicted and were not measured) or eggs.
- Concentrations in exposure media or tissue are measured, not estimated.
- Exposure duration is clearly reported, and effects of chronic exposures are evaluated.
- A standard or peer-reviewed study protocol is used.

Where high quality TRVs are not available, a conservative approach to developing points of comparison for exposure estimates is used. Estimated exposures falling below any of the TRVs selected indicate a conclusion of no risk with a high degree of confidence. If one or more studies generally meeting most of these acceptability criteria could not be identified for a given measurement endpoint, and estimated Site-specific exposures exceed the TRV, risks were described qualitatively and were discussed in the uncertainty section of the risk assessment report.

1.6 Methods Used for Aggregation of Toxicity Data

For most $COPC_{E}$s, reasonably conservative TRVs from the literature are compared directly to Site-specific exposure estimates. In these cases, the TRV reflects results of a study or studies of acceptable quality to provide the best representation of the receptor on the basis of taxonomy and sensitive life stages of the Site-specific receptor. If the estimated exposure is less than the individual TRV, no further analysis is conducted, and risk is considered negligible if both the central tendency exposure and reasonable maximum exposure are below the LOAEL ($HQ_{L} < 1$).

For dioxins, furans, and PCBs, there is a large body of literature describing toxicity to various species. Risks due to both 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) TEQs and total PCBs are considered. For these $COPC_{E}$s, several TRVs of equal quality and relevance were available in some cases. If fewer than 10 values were found, the following steps were taken to derive a TRV:

1. Within-species NOAELs or LOAELs were grouped.
2. The geometric means of the within-species NOAELs and the geometric mean of the within-species LOAELs were calculated.
3. Resulting geometric means within a TRV category (LOAEL or NOAEL) are pooled. No individual species is represented by more than one value, although some values are the results of only one study.
4. The geometric mean of the pool of data for multiple species is calculated, and that value becomes the NOAEL or the LOAEL for the $COPC_{E}$ and receptor.

The RI/FS Work Plan indicates that cumulative distribution functions derived from multiple effects-level metrics with a species, or species sensitivity distributions (SSDs), would be developed using multiple literature values for several species. This is a tool that can be used to clearly define the risk and the uncertainty associated with a risk calculation. However, sufficient data for a set of related taxa that have similar exposure and effects metrics were not found, except for the SSD for early life stage fish developed for Steevens et al. (2005).

A method to extrapolate TRVs between species on the basis of the difference in body weights between the two species, called allometric scaling, has been used at some Superfund sites. The technical basis for extrapolation of TRVs between species based on body size is not as well established for ecological receptors as it is for extrapolations relating to human health risk assessment (i.e., rat to human extrapolations), where it is most widely applied. Because of uncertainty in the use of allometric models to scale TRVs between species, particularly for birds, extrapolations on the basis of body size was not used to estimate or derive measures of effects when species-specific TRVs are not available.

2 ORGANIC COMPOUNDS

The organic chemicals considered $COPC_{ES}$ for one or more ecological receptors are dioxins, furans, PCBs, BEHP, carbazole, and phenol (Table B-1). Because dioxins and furans are the indicator chemical group at the Site, and because their toxicity is relatively well studied, greater depth of information is provided for them than for other organic $COPC_{ES}$. PCBs are also discussed at length because their toxicity in vertebrates can be considered additive with the toxicity of dioxins and furans, and the BERA evaluates risk using this additivity model.

2.1 Dioxins and Furans

Attachment B2 to the Screening Level Ecological Risk Assessment (SLERA) (Appendix B in Anchor QEA and Integral 2010) provides an overview of the technical literature available for the evaluation of dioxin and furan toxicity to birds, mammals, fish, reptiles, and invertebrates. The toxicity profile presented below repeats much of that information (e.g., for benthic macroinvertebrates) and provides a more focused summary and rationale for selection of studies that provide the basis for TRVs or SSDs to be used.

The main text of the BERA (Section 3.2) also repeats some of the basic information from Appendix B, Attachment B2 to the RI/FS Work Plan on the general toxicology of dioxins and furans, and the common conceptual framework used to present and evaluate dioxin and furan toxicity information for vertebrates. Readers are referred to that text for discussion of the basis for the use of TEFs and 2,3,7,8-TCDD TEQ concentrations or doses for assessment of exposure and toxicity to dioxins, furans, and dioxin-like PCBs. The approach used in this BERA is consistent with USEPA (2008) guidance.

From the perspective of an assessment of ecological risks, adverse effects of dioxins and furans on reproductive success, growth, and survival are relevant to evaluating the potential for population-level effects in any receptor. A range of reproductive and developmental effects such as reduced fertility, early-stage embryotoxicity, early life-stage mortality, developmental effects, and reduced growth of offspring are relevant, because these effects can conceivably affect the growth or viability of a population. This section provides a summary of information considered in the development of dioxin and furan TRVs for ecological receptors addressed by this BERA, identifies the TRVs to be used in the risk evaluation, and provides supporting rationale for their selection.

2.1.1 Benthic Macroinvertebrates

Several studies have found no adverse effects in aquatic invertebrates following exposure to TCDD; studies to provide systematic toxicity data for the other dioxin and furan congeners are rare. Data summarized in Attachment B2 to the SLERA include findings of no effects for the following:

- **Estuarine amphipods.** Barber et al. (1998) exposed the estuarine amphipod *Ampelisca abdita* to sediments spiked with 2,3,7,8-TCDD at concentrations between 1.1 and 25,000 ng/kg dw. No significant differences ($p > 0.05$) were found for survival or growth between any of the spiked sediments and the negative control sediments. Barber et al. (1998) identified an NOEC for 2,3,7,8-TCDD in sediment of 25,000 ng/kg for the amphipod.
- **Freshwater insect and oligochaete.** West et al. (1997) exposed the freshwater chironomid (*Chironomus dilutus*) and oligochaete (*Lumbriculus variegatus*) to nominal dietary concentrations of 30, 300, and 3,000 ng 2,3,7,8-TCDD/kg of total organic carbon (TOC) in food over 28-day (*Lumbriculus*) and 35-day (*Chironomus*) exposure periods. Maximum 2,3,7,8-TCDD body burdens in *Chironomus* and *Lumbriculus* were 5,084 and 9,533 µg/kg lipid at the 3,000 ng/kg TOC[2] treatment levels. No significant effects were found on survival, growth, or reproduction for either of the test species. Another study in which chironomids (*Chironomus riparius*) were exposed to 2,3,7,8-TCDD in spiked sediments with concentrations ranging from <150 to 10,000 ng/kg dw reported no significant effects on survival or growth and no significant differences in the occurrence of deformities from control (Loonen et al. 1996). The maximum observed tissue concentration was 14,000 ng/kg dw.
- **Marine polychaetes, musssels, and grass shrimp.** Pruell et al. (1993) and Rubenstein et al. (1990) evaluated the toxicity of 2,3,7,8-TCDD and 2,3,7,8-tetrachlorodibenzofuran (TCDF) on polychaetes (*Nereis virens*), bivalve molluscs (*Macoma nasuta*), and grass shrimp (*Palaemonetes pugio*) exposed to sediments collected from the Passaic River, New Jersey. The mean sediment concentrations of the two compounds were 656 and

[2] Measured concentrations were 3,804 ng/g TOC for *Chironomus* diet and 3,594 ng/g TOC for *Lumbriculus* diet.

334 ng/kg dw, respectively. The final tissue concentrations for the polychaetes, bivalves, and shrimp were 422, 142, and 138 ng/kg, respectively. There were no major differences in survival between the test and reference-area sediments, with control-adjusted survival of all three species being greater than 90 percent.

- **Freshwater zooplankton and snails.** Adams et al. (1986) evaluated the toxicity of 2,3,7,8-TCDD to daphnids (*Daphnia magna*) in 48-hour water exposures at concentrations of 0.2 to 1,030 ng/L, followed by a recovery period. The authors concluded that no toxic effects were found. Yockim et al. (1978) evaluated the toxicity of 2,3,7,8-TCDD to daphnids (*Daphnia magna*) and snails (*Helisoma* sp.) in 32-day water exposures at concentrations of 2.4 to 4.2 ng/L. The authors found no adverse effects on growth, reproduction, or feeding for either test species. Isensee and Jones (1975) evaluated the toxicity of 2,3,7,8-TCDD to snails (*Physa* sp.) and daphnids (*Daphnia magna*) in water exposures at concentrations of 0.05 to 1,300 ng/L. The authors concluded that no effects on reproductive activity, feeding, or growth were found for either test species.

One study reported adverse effects on freshwater crayfish (*Pacifastacus leniusculus*), a crustacean, following injections of 2,3,7,8-TCDD, but the results are not considered useful for risk assessment because of several uncertainties (Ashley et al. 1996). Although mortality was observed, the authors concluded that the cause of mortality was not associated with tissue pathology, and could not specify the cause of the observed mortality. In addition, the authors acknowledged that their sample sizes were small, with only three to six crayfish exposed to each 2,3,7,8-TCDD concentration in three separate experiments. Finally, the methods indicate that excessive dimethylsulphoxide may have been used in dosing solutions, which could have contributed to mortality.

The available published studies on this topic are summarized in Attachment B2 to the SLERA (Anchor QEA and Integral 2010), included here as Table B-4. This compilation of literature and related analysis finds that, in contrast to fish and wildlife, most studies of aquatic invertebrates have found that invertebrates are relatively insensitive to TCDD toxicity. Although aryl hydrocarbon receptor (AhR) homologues have been identified in various invertebrate species, invertebrate AhR homologues lack the ability to bind dioxins (Hahn et al. 1992; Butler et al. 2001). However, recent studies have documented reproductive toxicity of 2,3,7,8-TCDD in bivalve molluscs. The mechanism by which dioxins affect bivalve molluscs has yet to be identified with certainty, but researchers agree that it is independent of AhR homologues.

The only series of studies of acceptable quality (Section 1.4) showing effects of 2,3,7,8-TCDD on invertebrates involve injection of TCDD into the eastern oyster (*Crassostrea virginica*) and the soft-shell clam (*Mya arenaria*), both bivalve molluscs. They found that 2,3,7,8-TCDD preferentially accumulates in the gonads and digestive glands of the bivalves, which is consistent with the earlier findings of Rhodes et al. (1997). They speculated that this uptake was not solely related to lipid content, but was "best explained by 2,3,7,8-TCDD binding to a tissue-specific receptor" (Wintermyer et al. 2005).

Cooper and Wintermyer (2009) found a time-dependent loss in body mass and microscopic abnormalities in multiple tissues in clams following a single exposure administered by gavage of 200 ng 2,3,7,8-TCDD/kg tissue ww, but no loss in body mass following 24-hour waterborne exposure or muscle injection to achieve the same tissue concentration. Oysters exhibited decreases in gonadal development in females at tissue concentrations of 2.0 ng/kg, which is consistent with the sensitivity found by Wintermyer and Cooper (2003, 2007). Although gonadal development in the clams was not evaluated by Cooper and Wintermyer (2009), Butler et al. (2004) found a lack of proper gonadal development in both female and male clams at comparable 2,3,7,8-TCDD concentrations. Wintermyer and Cooper (2007) found a shift in the male/female ratios for the oysters, with a decrease in the number of females, at tissue concentrations of 10 ng/kg (Table B-4).

Although the use of injection and gavage may not precisely mimic processes of exposure and uptake in environmental settings, these recent studies provide useful information, updating the literature on the toxicity of 2,3,7,8-TCDD to some invertebrates. Cooper and Wintermyer (2009) conclude that their data, together with the studies they reviewed, provide evidence for sensitivity of reproductive endpoints in bivalve molluscs to 2,3,7,8-TCDD exposure, and that tissue concentrations that resulted in altered gonadal development and reduced larval survival in the laboratory (i.e., 2 to 10 ng/kg) were comparable to the levels observed in field populations of *M. arenaria* (4.8 to 20 ng/kg) and *C. virginica* (0.15 to 3.2 ng/kg) in chemically contaminated waterbodies in New Jersey (i.e., Newark Bay, Arthur

Kill, and Sandy Hook), where bivalves are known to be stressed. Cooper and Wintermyer (2009) concluded that 2,3,7,8-TCDD alters normal development of reproductive organs and larval development in tested bivalves at whole-organism tissue concentrations of 2 to 20 ng/kg, although they acknowledge that the estuaries they evaluated were affected by numerous chemicals other than 2,3,7,8-TCDD.

However, Cooper and Wintermyer (2009) draw the conclusion, that survival of oyster larvae is impaired by TCDD at 2 ng/kg tissue from their 2003 field study (Wintermeyer and Cooper 2003). This may overstate the role of TCDD in survival of larvae. Wintermyer and Cooper (2003) transplanted wild-caught adult eastern oysters (*Crassostrea virginica*) to Newark Bay, the Arthur Kill area of Raritan Bay, and Sandy Hook, New Jersey. Results suggest that oysters with TCDD (ng/kg)/TCDF (ng/kg)/total PCB (μg/kg) concentrations of 3.2/2.1/68 and of 1.3/1.7/65 had reduced survival of veliger larvae. Conditions of this study are not analogous to conditions at the SJRWP site because of the relatively high levels of PCBs in the oyster tissue, which could have been the cause of reductions in larval survival, found in combination with the TCDD levels that are reported. Also, Wintermyer and Cooper (2003) exposed test organisms in complex urban estuaries, where sediment and water quality are influenced by oil refineries, urban runoff, combined sewer overflows, sewage treatment plants, and other sources of anthropogenic pollutants. The effects of estrogenic compounds and other chemicals in addition to TCDD, TCDF, and PCBs were not considered or discussed by Wintermyer and Cooper (2003), and exposures of test organisms to other chemicals were not evaluated. However, Wintermyer and Cooper (2003) also exposed oysters to TCDD alone in a controlled experiment, and found reduced larval survival at the lower tissue concentration (2 ng/kg ww). Therefore, although the field study cannot account for the effects of the mixtures, the laboratory study demonstrates that 2 ng/kg ww in whole eastern oyster tissue causes reduced fertilization and reduced larval survival in eastern oysters.

According to the earlier analysis of this information (Attachment B2 to the SLERA), the assessment of risks associated with dioxin exposures to molluscs will be based on the assumption that bivalve molluscs are among the most sensitive invertebrate taxa, and that evaluations based on toxicity of TCDD to bivalves are protective of benthic macroinvertebrates as a group. Therefore, the TRVs for performing the risk evaluation for the benthic macroinvertebrate community and for bivalves are as follows:

- **Benthic macroinvertebrate community.** A no-observed-adverse-effect concentration (NOAEC) only, calculated as the geometric mean of survival NOAECs reported in spiked sediment bioassays summarized in Table B-4, will be used in the BERA. The geometric mean of these NOAEC values is 2,343 ng/kg dw (Barber et al. 1998; Pruell et al. 1993; Rubenstein et al. 1990; Loonen et al. 1996).
- **Bivalves.** The studies summarized by Cooper and Wintermyer (2009) indicate that tissue concentrations in the range of 2 to 20 ng/kg may cause adverse reproductive and developmental effects in bivalve molluscs (Table B-4). The lowest-observed-adverse-effect concentration (LOAEC) of 2 ng TCDD/kg ww tissue for delayed gonadogenesis in males and histopathology in females (Wintermyer and Cooper 2007) and reduced egg fertilization and larval survival (Wintermyer and Cooper 2003) was chosen for use in the BERA. A corresponding NOAEC was not available.

Although Table B-4 presents NOAECs and LOAECs using a variety of metrics, the approach for the benthic macroinvertebrate community was selected because sediment concentrations have been empirically measured at the Site and in the supporting studies, eliminating any need for modeling. Also, the studies supporting derivation of the NOAEC above span several major taxonomic groups, including arthropods, crustaceans, annelids, and molluscs. In light of the absence of effects on most other invertebrate taxa at tissue concentrations greater than the effects levels reported by Cooper and Wintermyer for clams and oysters, the use of bivalve molluscs as a surrogate benthic invertebrate taxon for evaluating the exposure and potential effects of TCDD on benthic macroinvertebrates generally at the Site would be an overly conservative means to address risk to benthic macroinvertebrates as a group. Results of comparisons of concentrations in clam tissue to the CTRs from studies with clams and oysters are therefore considered applicable only to assessment of risks to bivalves.

2.1.2 Fish

Some fish species appear to be among the most sensitive of vertebrates to dioxin and furan toxicity and are thought to be the most sensitive of aquatic taxa (USEPA 2008). Dioxin toxicity in fish is mediated via the AhR pathway, as it is in birds and mammals. Dioxins, individual 2,3,7,8-substituted congeners, and mixtures of dioxin-like compounds produce similar early life-stage toxic effects in fish, supporting the conclusion that toxicity is mediated through a common mechanism (Walker et al. 1996). Unlike mammals, which possess a single form of AhR, fish can have multiple AhR homologues, possibly due to a gene duplication event that occurred during the evolution of fish species (Carney et al. 2006; Andreasen et al. 2007). It is not yet clear what role the different homologues play in dioxin toxicity to fish (USEPA 2008), but this information suggests the possibility of substantial variation in sensitivity among fish species.

As for other ecological receptors, early life stages are the period of greatest sensitivity of fish to dioxin toxicity (Walker and Peterson 1991; Elonen et al. 1998; Steevens et al. 2005; Carney et al. 2006). Multiple fish species, including brook trout, catfish, northern pike, fathead minnow, zebrafish, and medaka, have been shown to be particularly sensitive to dioxin toxicity during the life stages from hatching to swim-up (following absorption of the yolk sac and transition into the water column for feeding) (Walker and Peterson 1991; Elonen et al. 1998). Toxic effects to the egg are seen during later embryonic development as well. For example, embryonic zebrafish exposed to 2,3,7,8-TCDD within a few days post-fertilization begin to manifest toxic responses, including pericardial and yolk sac edema (described further below), reduced cardiac function, and alterations to cartilage growth at 48 to 120 hours post-fertilization, when morphogenesis of primary organ systems and embryo growth are occurring (Carney et al. 2006).

Studies exposing post-swim-up trout fry to concentrations of dioxins associated with significant increases in mortality in the pre-swim-up fry did not find significant mortality in the later life stages (Walker and Peterson 1991; Walker et al. 1996). Possibly because the importance of early life stage toxicity in fish was established relatively early, sublethal effects of dioxins and furans on juvenile and adult fish, including potential effects on feeding, growth, predator avoidance, and other functions important to fish survival and reproduction, are not as well studied (Carney et al. 2006). It is also notable that the literature suggests that population resistance to dioxin toxicity can also occur over time in some fish, as shown for a killifish population living in the vicinity of a Superfund site with high dioxin levels (Nacci et al. 2002).

2.1.2.1 Reproductive Effects

Effects on reproduction, including decreased egg productivity (number of eggs produced per female) and decreased spawning success (production of eggs that are successfully fertilized) have been observed in fish following chronic exposures in experiments with dioxins that result in tissue concentrations in the range of nanograms per gram. No effects on fertility were found in adult brook trout exposed for 28 days to a range of dioxin concentrations targeted to achieve 0, 75, 150, 300, 600, and 1,200 ng/kg adult tissue (which achieved a concentration range of up to 517 ng/kg egg tissue through maternal transfer) (Johnson et al. 1998; Tietge et al. 1998). However, chronic dietary exposure of adult female zebrafish to an estimated dose of 0, 80, 320, or 800 pg TCDD/day for 20 days (corresponding to measured body burdens of 0, 1,100, 6,900, and 15,000 ng/kg at the end of the exposure period) led to significant adverse reproductive effects in the two highest exposure groups, including decreased egg production. At the highest exposure, a reduction in the number of ovarian follicles and decreased spawning success of up to 80 percent relative to control were observed (Heiden et al. 2009).

2.1.2.2 Developmental Effects

Developmental effects are manifested in early life stages during critical developmental processes in embryonic and newly hatched fish. They are often symptomatically similar to "blue sac disease," a disease of yolk sac fry that was first characterized as a response to poor conditions in hatcheries. Blue sac disease is characterized by edema, or liquid accumulation, in the yolk sac, causing swelling, which can lead to reduction or destruction of circulatory function in the yolk sac and/or body. Experimental exposure of fish eggs to 2,3,7,8-TCDD has led to increased incidence of symptoms very similar to blue sac disease, including subcutaneous edema with loss or cessation of blood circulation in the yolk sac and body (Spitsbergen et al. 1991; Carney et al. 2006).

Additional effects of dioxin exposure at the egg stage that manifest after hatching and prior to swim-up can include necrosis of the retina, brain, and spinal cord, malformations of the tail fin, microcephaly, and deformities of mandibular and opercular bones (Spitsbergen et al. 1991; Elonen et al. 1998; Johnson et al. 1998). Johnson et al. (1998) noted symptoms of edema in brook trout fry at lower exposure concentrations (84 ng/kg egg) than concentrations associated with significant incidence of opercular and mandibular deformities (156 ng/kg).

Cardiovascular dysfunction is regarded as an important adverse effect of 2,3,7,8-TCDD in fish, and is particularly well studied in zebrafish, which are frequently used as an animal model for 2,3,7,8-TCDD toxicity (Carney et al. 2006). In addition to pericardial edema, toxic effects of 2,3,7,8-TCDD on cardiovascular development in fish include inhibition of growth and normal development of the common cardinal vein, a paired vessel that grows across the yolk, connects to the heart, and is extensively reorganized during later embryonic development (Bello et al. 2004). 2,3,7,8-TCDD exposure was observed to cause physiological alteration in atrioventricular and bulboventricular valve development of the zebrafish, leading to an inability of the heart to function effectively in circulating blood (Mehta et al. 2008). Although zebrafish have been the most intensively studied species with respect to mechanisms of dioxin effects on cardiotoxicity, cardiac effects of dioxin exposure, including pericardial hemorrhage and myocyte necrosis in trout fry exposed to 2,3,7,8-TCDD have been shown for other fish species as well (Spitsbergen et al. 1991). Linking cardiovascular effects in fish to ecological endpoints in the BERA is not straightforward and is likely prohibitively uncertain in a BERA context. Nevertheless, the available information on cardiovascular effects is useful in understanding the ways dioxins may affect wild fish at the Site.

2.1.2.3 Effects on Growth

Sublethal effects of 2,3,7,8-TCDD exposure can include malformations of cartilage in the developing fish, leading to reductions in length of or alterations to parts of the skeletal structure (Spitsbergen et al. 1991; Carney et al. 2006). Growth, as measured by body length or weight in juveniles, was reduced in white sucker exposed at the egg stage to 2,3,7,8-TCDD resulting in a tissue concentration of 1,220 ng 2,3,7,8-TCDD/kg egg tissue and in lake herring at 717 ng/kg egg tissue (Elonen et al. 1998). Elonen et al. (1998) further suggested that reduced lengths observed in fish exposed to 2,3,7,8-TCDD were related to the manifestation of edema, leading to prevention of blood flow through the yolk sac vasculature, and ultimately resulting in decreased absorption of nutrients to the body.

2.1.2.4 Development of a Species Sensitivity Distribution for Fish

Among tested freshwater fish species, sensitivity to 2,3,7,8-TCDD-induced early life stage toxicity ranges approximately 50-fold, with salmonids being the most sensitive and zebrafish the least sensitive species (Walker and Peterson 1991; Elonen et al. 1998; USEPA 2008). Steevens et al. (2005) compiled data on effects of dioxins and furans to embryos from 10 studies of several fish species, generating a summary of the geometric means of NOAEC and LOAEC sublethal growth and reproduction endpoints ranging from 0.42 µg/kg lipid for lake trout to 60.28 µg/kg lipid for zebrafish. They also compiled lethal effects (LR_{50}) concentrations ranging from 0.53 µg/kg lipid for lake trout to 153.53 µg/kg lipid for zebrafish. The similarity in the ranges of the sublethal and lethal effect concentrations reflects the steep dose-response associated with dioxin toxicity in fish (i.e., the transition from a concentration that causes an observable sublethal effect to a concentration causing a lethal effect occurs over a small range). For some salmonids (e.g., brook trout and lake trout), this transition occurs within less than a 1 ng/g increase in concentration (Elonen et al. 1998; Steevens et al. 2005).

Steevens et al. (2005) fitted the fish egg tissue residue data to a logistic distribution to generate an SSD based on the geometric mean of the LOAEC and NOAEC for all 10 studies, and a second one using the LC_{50} values (i.e., concentrations lethal to half the test organisms). The 10 geometric means (Table B-5) were used to generate an SSD for fish exposed to TCDD and dioxin-like compounds. Using the resulting SSD, Steevens et al. (2005) generated egg tissue residue-based TRVs for dioxin-like compounds in fish tissue that are protective of specified percentiles (e.g., 95, 97.5, and 99 percent) of species, with confidence limits. The SSD developed by Steevens et al. (2005) is used to evaluate effects of dioxin and furan exposures in fish for this BERA. By necessity, this risk assessment uses TEQ_F concentrations in whole body samples of fish for comparison to the CTRs of Steevens et al. (2005). This approach conservative. Tietge et al. (1998) found that TCDD concentrations in eggs of brook trout (*Salvelinus fontinalis*) were just 39 percent of the concentrations in the whole fish. Heiden et al (2005) reported an even lower level of egg accumulation of TCDD relative to female whole bodies in zebrafish, with egg concentrations of just 5 percent of whole adults. This risk assessment is conservative because it assumes a 1 to 1 ratio of whole adult fish to egg concentrations.

Steevens et al. (2005) remark that because so many of the species represented in the SSD are salmonids, which are generally very sensitive to many toxicants, the resulting toxicity residue benchmarks derived from the SSD are conservative for many non-salmonid fish species. There are no fish receptor surrogates for the SJRWP that are salmonids, so the SSD derived by Steevens et al. (2005) is considered to be conservative for application at the Site.

2.1.3 Reptiles

Limited data are available regarding the toxicity of dioxin-like compounds to turtles, and the potential effects of dioxins and furans on other reptiles have not been studied (Portelli and Bishop 2000). Studies with turtles have generally been conducted in the field, where exposures to other chemicals may occur, so conclusions about the effects solely from exposure to dioxins and furans are not possible. The available data for turtles suggest that additional controlled studies of adverse effects are needed to understand dioxin and furan toxicity in this taxon. The available studies are summarized below.

2.1.3.1 Ethoxyresorufin-O-Deethylase Induction in Snakes

Liver cells from the African brown snake (*Lamprophis fuliginosus*) were exposed *in vitro* to TCDD and four non-ortho substituted co-planar (i.e., dioxin-like) PCB congeners (PCB77, PCB81, PCB126 and PCB169) (Hecker et al. 2006). Dose-dependent increases in ethoxyresorufin-O-deethylase (EROD) activity were observed with exposure to TCDD and PCB126, but not with the other PCB congeners, suggesting lower sensitivity of this snake than other vertebrates to the dioxin-like toxicity of PCBs. The potency in EROD induction by PCB126 relative to TCDD was comparable to potency in mammals and the more sensitive birds, but indicated a higher sensitivity of the snakes than of fish. This information cannot be used to interpret estimated exposures in the field.

2.1.3.2 Reproductive Effects

Bishop et al. (1991) documented an increase in unhatched eggs and deformities in snapping turtles collected from an area contaminated with multiple potential toxicants, including PCBs, dioxins, and furans. Because the exposure was to a mixture, it is not possible to attribute the effects to one or a specific subgroup of the chemicals measured.

In a separate study (Bishop et al. 1998), turtle eggs exposed to a mixture of dioxins, furans, and PCBs (as well as pesticides and other chemicals) collected from the field had significantly increased proportion of unhatched eggs and increased proportion of deformed hatchlings. However, data from this study are limited in their usefulness, because of the presence of multiple contaminants and because correlation between individual contaminants and adverse outcomes was not conducted. Portelli and Bishop (2000) indicate that the rates of abnormalities observed in these two studies correlated with dioxin, furan, and PCB concentrations in eggs, but not when they were expressed as TEQ.

More recently, a field study and a controlled experiment addressing the potential for hormonal alterations and reproductive effects in turtles have been conducted. Evaluation of plasma hormone levels in male yellow-blotched map turtles collected from a TCDD-contaminated area revealed that estradiol increased and testosterone decreased in a small proportion of turtles (Shelby and Mendonça 2001). The potential reproductive consequences of the observed changes in hormone levels are not clear. Although some evidence of turtle sex reversal has been observed following exposures of turtles to PCBs (Willingham et al. 2000), raising concerns that this might be a result of dioxin-like toxicity, 2,3,7,8-TCDD has been shown not to cause sex reversal following administration to eggs in a laboratory study (Gale et al. 2002). Sexual development of reptiles is also linked to incubation temperature (e.g., Nichols et al. 1999), in the absence of chemical contamination, suggesting that toxicological endpoints such as those listed above be interpreted with caution.

Quantitative measures of effects are not available for evaluation of the risks of exposures of reptiles to dioxins and furans at the Site. A quantitative estimate of ingestion exposure to snapping turtles is planned, and this can be used to evaluate the reptile exposures relative to those of other receptors. In the absence of new information by the time the BERA is drafted, risks to reptiles will be evaluated qualitatively.

2.1.4 Birds

Evaluation of the toxic effects of dioxins and furans has been conducted in multiple North American bird species, including herons, egrets, terns, cormorants, a bluebird species, chickens, pheasants, and ducks. Toxicity of dioxins and furans in birds is mediated through

AhR, and may also occur through other biochemical pathways. Endpoints evaluated in birds have included reproductive success (using a variety of endpoints), thyroid toxicity, cardiovascular toxicity, immune toxicity, and effects on growth and survival.

As for fish, AhR-mediated toxicity is the focus of this toxicity profile for birds because AhR-mediated effects are assumed to occur at lower doses than other effects. Also, effects on early life stages, which are well documented for birds exposed to dioxins, furans and dioxin-like PCBs (e.g., Henshel et al. 1997), are also emphasized because they are relevant for understanding risks to bird populations.

As described in Appendix B, Attachment B2 to the RI/FS Work Plan (Anchor QEA and Integral 2010), two lines of evidence are used in the BERA to evaluate risks to birds from exposures to dioxins and furans: comparison of estimated daily ingested doses to TRVs expressed in the same terms (mg/kg bw-day), and comparison of estimated egg concentrations to TRVs expressed in the same terms (ng TEQ/kg egg ww). This section does not repeat the information presented in Attachment B2; instead it expands on that discussion, focusing on data supporting development of TRVs for these two lines of evidence.

2.1.4.1 Variability in Avian Toxicity of Dioxins and Furans

Exposure to dioxin-like compounds is associated with both embryo mortality and a variety of adverse effects on chick development, including reduced embryo and hatchling growth, deformities, and abnormalities in the developing heart. The exposure levels at which adverse effects are observed span a large range across avian taxa, from low nanograms per kilogram to low micrograms per kilogram in tissue.

There are clear differences among bird species in susceptibility to dioxin-like toxicity, which have been attributed to biochemical differences in AhRs among species (Karchner et al. 2006; Head et al. 2008). Domestic chickens are generally considered to be the most sensitive bird species tested, not only in responses to TCDD exposure, but also in responses to other dioxin-like compounds, such as TCDF and PCB126 (TN & Associates 2002). Even the sensitivity among chicken species measured as EROD induction (by PCB 126) varies by a factor of 20, with chicken species having greater sensitivity than species with wild populations. The next most sensitive species (as measured by EROD induction) is the pheasant (TN & Associates 2002), followed by turkeys, ducks, and gulls. Using a different set of assays, common terns were significantly less sensitive to TCDD toxicity than chickens. Further, according to TN & Associates (2002), an experiment by Sanderson et al. (1998) compared EROD induction in bird hepatocyte cultures for several species. Among other things, this study reports 40-fold and 80-fold variation in $EC_{50}s$ for EROD induction in TCDD-exposed hepatocytes of ring-billed gulls and double crested cormorants, respectively. Although intraspecies sensitivity is not often discussed, these studies demonstrate that species-specific differences are relevant to understanding ecological risks.

The general finding that chickens are most sensitive has been verified for EROD induction and egg mortality, but is less clear for embryo developmental endpoints. Cohen-Barnhouse et al. (2011) found that changes in developmental endpoints in embryos of chickens, pheasants, and quail were not consistently related to dose of TCDD, TCDF, or 2,3,4,7,8-pentachloro-dibenzofuran (PeCDF). Although their results generally supported the view that chickens are the most sensitive for egg mortality, developmental abnormalities occur at different stages following egg laying, and these effects did not follow simple dose-response relationships. Cohen-Barnhouse et al. (2011) report that post-hatch mortality of surviving chicks under all treatments did not differ from that of the vehicle control. Bruggeman et al. (2005) evaluated reproductive performance of domestic chicken hens that were exposed to TCDD *in ovo*, and although physiology was affected, reproductive performance was not. Because of these findings, this toxicity profile focuses on studies in which egg mortality is the endpoint.

2.1.4.2 Reproductive Effects and Toxicity to Embryos

Many studies have been conducted to address the toxicity of TCDD and dioxin-like PCBs to bird eggs and developing embryos, but not all of the available studies can be used for risk assessment (Section 1.4). Given the complexity of the literature describing avian toxicity in terms of the variety of species, the ranges of results, and the methods used for selection of studies, USEPA (2003) was used as a starting point for the literature evaluation needed to identify TRVs expressed as egg concentrations for this BERA. The literature presenting information for use in developing ingestion TRVs is much smaller. Both are discussed below, as are the selected TRVs.

2.1.4.2.1 Egg Tissue TRVs

Use of egg-exposure based TRVs is the recommended risk assessment approach by both TN & Associates (2002) and USEPA (2003). USEPA (2003) provides a compilation of results of toxicity tests in which exposures as concentrations in eggs were documented, building on the detailed literature review conducted by TN & Associates (2002) for USEPA's Office of Research and Development. They used their aggregation of data to prepare SSDs for birds. Both laboratory and field studies were compiled by USEPA (2003). A paper was only selected for use in USEPA's (2003) analysis if it included all of the following:

- Evaluation of more than one quantitative dose or exposure level. Studies evaluating only one dose or exposure level were considered to have too much uncertainty.
- One or more quantifiable toxicological endpoint.
- Appropriate statistical tests showing significant changes in response with changes in dose or exposure levels.
- Evaluation of the potential for co-contaminants to affect results (for field studies).

USEPA's (2003) compilation of TRVs expressed as TCDD (or TEQ) concentrations in eggs includes NOAELs for developmental impairment from laboratory studies ranging from 66 ng TEQ/kg egg for the chicken to 50,000 ng TEQ/kg egg for several other bird species, including two gull species, the Graylag goose, and the goldeneye (a duck). Corresponding LOAELs range from 150 to 4,400 ng TEQ/kg egg. Not all of these studies were used for developing egg tissue TRVs, as discussed below.

Finally, TCDD or other toxicants can be injected into bird eggs in one of several ways: into the air cell in the egg, into the albumin (white) of the egg, or into the yolk. Nosek et al. (1992a) injected radiolabeled TCDD into laying pheasant hens once a week for 10 weeks; hens were induced to lay eggs during the final two weeks of exposure. The first 15 eggs from each hen were collected, and the yolk and albumin were separated. Nosek et al. (1992a) report that greater than 99 percent of the TCDD-derived radioactivity translocated from hens to eggs was found in the yolk; none was detected in albumin. Some authors have noted that injection into the air cell may contribute to egg suffocation, when the oily material used in dosing impedes the transfer of oxygen to the embryo (Henshel et al. 1997a). For these reasons, TRVs reported by laboratory studies in which dosing occurred by injection into the yolk are preferred because they are considered to be more biologically realistic.

To summarize, controlled laboratory studies meeting USEPA (2003) criteria and using injection into yolks or maternal transfer as the means of administration and in which egg mortality was the endpoint were preferred for this risk assessment. Although there are a number of studies of the toxicity of PCB126 and other PCB congeners to birds, because of the inter- and intraspecies variability, and uncertainties about Van den Berg et al.'s (1998) TEFs for birds (e.g., Cohen-Barnhouse et al. 2011), Integral also used only studies in which TCDD was the toxicant of interest.

USEPA (2003) cites data from the following studies of TCDD toxicity in its compilation of egg tissue TRVs, which was the starting point for this evaluation:

- Henshel et al. 1997a. The Relative Sensitivity of Chicken Embryos to Yolk- or Air-Cell-Injected 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin.
- Powell et al. 1996. Effects of 3,3',4,4',5-Pentachlorobiphenyl (PCB 126) and 2,3,7,8-Tetrachlorodibenzo-*p-D*ioxin (TCDD) Injected into the Yolks of Chicken (*Gallus domesticus*) Eggs Prior to Incubation.
- Powell et al. 1997a. Effects of 3,3',4,4',5-Pentachlorobiphenyl (PCB 126), 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD), or an Extract Derived from Field-Collected Cormorant Eggs Injected into Double-Crested Cormorant (*Phalacrocorax auritus*) Eggs.
- Powell et al. 1997b. Organochlorine Contaminants in Double-Crested Cormorants from Green Bay, Wisconsin: II. Effects of an Extract Derived from Cormorant Eggs on the Chicken Embryo.
- Powell et al. 1998. Effects of 3,3',4,4',5-Pentachlorobiphenyl and 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin Injected into the Yolks of Double-Crested Cormorant (*Phalacrocorax auritus)* Eggs Prior to Incubation.
- Nosek et al. 1992b.[3] Toxicity and Reproductive Effects of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin in Ring-Necked Pheasant Hens.

[3] USEPA (2003) cites Nosek et al. (1992b), but the data appear to be from Nosek et al. (1993). Integral used results from both studies.

- Henshel et al. 1997b. Brain Asymmetry as a Potential Biomarker for Developmental TCDD Intoxication: A Dose-Response Study.
- Walker et al. 1997. Expression of the Aryl Hydrocarbon Receptor (AhR) and AhR Nuclear Translocator during Chick Cardiogenesis Is Consistent with 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin-Induced Heart Defects.

Henshel et al. (1997b), Powell et al. (1997b), and Walker et al (1997) were not used because the study endpoints were not relevant to evaluation of ecological risks. Integral also reviewed and used data from two additional studies exposing eggs via yolk injection or maternal transfer by Nosek et al. (1992a; 1993). Results of three studies in which the TCDD was administered via injection into the air sac or albumin are also summarized to provide valuable perspective on the yolk injection studies: Cohen-Barnhouse et al. (2011), Henshel et al. (1997a), and Nosek et al. (1993).

Finally, the majority of literature on dioxin toxicity to birds reports information on field collected eggs. USEPA (2003) also compiled results from field studies and analyzed them separate from laboratory studies. Results of field studies are uncertain because of potential for effects of co-contaminants including pesticides, PCBs, and chemicals not measured by investigators, and therefore most field studies are not appropriate for use in risk assessments to define effects levels. However, no-effects levels derived from field studies can provide some perspective on Site-specific exposure estimates. A subset of the available field studies and the NOAECs for eggs that they report is also summarized.

Laboratory Based Yolk-Injection Studies

Studies by Nosek et al. (1992a,b; 1993) and Powell et al. (1997a, 1998) on common pheasants and double crested cormorants in which yolks were injected with TCDD form the basis for the TRV for bird eggs used in this risk assessment. Their results are summarized in Table B-6, along with two chicken studies included for perspective. These studies can be summarized as follows:

- Nosek et al. (1992b) conducted experiments with ring-necked pheasant hens, dosing individual hens weekly with intraperitoneal injections of 0, 0.01, 0.1, and 1.0 µg TCDD/kg bw per week. Significant reductions in survivorship, reduced egg production and an increase in cumulative egg mortality were observed in hens receiving 1.0 μg/kg-week, or a cumulative dose of 10 μg/kg bw, but not at lower doses. The effect observed at this dose was 100 percent egg mortality. Nosek et al. (1992b) estimate that the cumulative dose to individual hens resulting in 50 percent egg mortality was 4.5 μg/kg bw. This value and data from two other studies were used to derive the egg TRVs from the Nosek et al. (1992b) study: In the first study, Nosek et al. (1992a) injected 0.1 μg/kg bw radiolabeled TCDD into pheasant hens once a week for 10 weeks and found that about 1 percent of the total dose to hens was translocated to each of the first 30 eggs laid. Thus, the approximate dose to eggs resulting in 50 percent egg mortality (from Nosek et al. 1992b) is 45 ng/egg. In the second study, Nosek et al. (1993) report a mean egg weight for their pheasants of 30.5 g. This information was used to calculate a LOAEL for eggs from the Nosek et al. (1992b) of 1,477 ng/kg ww egg. The NOAEL from Nosek et al. (1992b) was similarly derived using a cumulative dose to hens of 1 μg/kg bw, resulting in 10 ng/egg or 328 ng/kg ww (Table B-6).

- Nosek et al. (1993) injected TCDD into yolks of pheasant eggs (and into albumin, below) at doses of 0, 10, 100, 1,000, and 10,000 ng /kg egg on Day 0 of embryonic development. An increase in mortality over the control group of 20 percent of eggs was observed at 1,000 ng/egg, and 98 percent mortality was observed at the highest dose. For the purposes of risk assessment, 1,000 ng/kg egg is considered the LOAEL from this study. The LD_{50} (the egg concentration at which 50 percent of organisms die) was calculated as 2,150 ng/kg egg.
- Powell et al. (1997a; 1998) conducted studies using eggs of double-crested cormorant collected from a remote area of Canada. The authors acknowledge the presence of TCDD and PCBs in cormorant eggs from that region, but performed a test to see whether these baseline residues would influence their studies. They injected an extract of TCDD and PCBs from untreated eggs into test eggs (Powell et al. 1997a) and found no effect on egg mortality, so the baseline level of contaminations is not likely to interfere with the experiments involving higher doses. In the 1997 study, these authors injected yolks of cormorant eggs with 60, 250, 1,000 and 4,000 ng TCDD/kg egg. Egg mortality was significantly elevated over controls only at the highest dose. In the 1998 study, egg yolks were injected with 1,300, 5,400, 10,700 and 11,700 ng/kg egg. At concentrations of 5,400 ng/kg and higher, mortality was significantly elevated

over controls. However, even at the highest dose, about 15 percent of eggs survived, demonstrating the low sensitivity of cormorants to TCDD relative to other species. This series of studies has been criticized for high control mortality, but such is to be expected with wild-captured eggs. The authors demonstrated that low levels of TCDD and PCBs in the collected eggs had no effect on the outcome, and even if it did, the result would be a conservative TRV. Otherwise, these studies are robust, demonstrating clear dose-response relationships, and are considered valuable because the eggs were from a wild stock.

Data for these four laboratory studies form the basis for derivation of the TRV for bird eggs used in this risk assessment (Tables B-6 and B-7). There were insufficient data for derivation of an SSD for bird eggs. The geometric means of NOAELs and LOAELs from the two studies for each of the two species were calculated, and the geometric mean of the resulting two geometric means were calculated to derive the TRVs for this risk assessment. The resulting NOAEL and LOAEL for TEQ_B in bird eggs, rounded to two significant figures are 450 and 2,400 ng/kg egg, respectively. The within-species geometric means were calculated first to minimize the influence of any one species on the final TRVs.

Because domestic chickens are clearly more sensitive than all other species, results from studies with chickens were not included in derivation of the TRVs. However, there are two yolk injection studies with domestic chickens summarized in Table B-6:

- Henshel et al (1997a) injected yolks of chicken eggs with 10, 30, 60, 100, 300, and 1,000 ng/kg egg ww and observed significantly elevated mortality (100 percent) at 300 ng/kg. The sample size in this study was fairly small per treatment, but a dose-response relationship was observed.
- Powell et al. (1996) also injected eggs of domestic chickens at 0, 40, 80, 160, 320, and 640 ng/kg egg. A statistically significant increase in egg mortality was observed at 160 ng/kg.

Results of these two studies are included in Table B-6, and the geometric mean NOAEL and LOAEL were calculated to be 89 ng/kg and 219 ng/kg, respectively. Using all three geometric means (chickens, pheasants, and cormorants) yields an overall geometric mean NOAEL rounded to two significant figures of 260 ng/kg and LOAEL of 1,100 ng/kg for bird eggs. These are considered in the uncertainty assessment to evaluate exposure estimates for birds.

Laboratory-Based Albumin or Air Cell Injection Studies

Studies in which eggs are exposed via injection to the albumin or the air sac are summarized in Table B-8. Results generally agree with those of the yolk injection studies, two of which are discussed above (Henshel et al. 1997a; Nosek et al. 1993), although chickens seem somewhat less sensitive using these data. In the third, Cohen-Barnhouse et al. (2011) was a detailed study evaluating relative sensitivity among bird species to developmental effects. The three bird species they studied are considered to have widely different sensitivities on the basis of enzyme induction. While this study presents a lot of important information, including an indication that 2,3,4,7,8-PeCDD and TCDF are more toxic to pheasants and quail eggs than are other congeners, this summary is focused on egg mortality. Results indicate that quail are considerably less sensitive than the other bird species discussed in this toxicity profile, with a NOAEL for egg mortality of 3,542 ng/kg, and an LOAEL of 9,015 ng/kg. These data were not included in calculation of geometric means because they are not yolk-injection data, but suggest that the TRVs derived for this risk assessment are reasonably conservative, in light of uncertainties.

Field Studies

In field studies reviewed by USEPA (2003), NOAELs for developmental effects range from 5 ng TEQ/kg egg for the wood duck to 1,440 ng TEQ/kg egg for the Caspian tern. Among all of the studies available on this topic, this summary is a selection of recently published literature including one from the area of the Site, for species selected as receptors for this risk assessment, and studies in which the authors performed a risk assessment for birds. The following studies are summarized in Table B-9:

- Frank et al. (2001) evaluated concentrations of multiple persistent organic compounds in waterbird eggs in the Galveston Bay area. In addition to several areas sampled within the Galveston Bay area, two reference areas were included for comparison of chemical levels in eggs and adverse health effects. Eggs were collected from three bird species: neotropic cormorants (n=28 eggs from four sites; n=18 eggs from two reference sites), black-crowned night herons (n=9 eggs from one site), and great egrets (n=7 eggs from one site). The collected eggs were evaluated for concentrations of pesticides, PCBs, dioxins, and furans. Egg extracts were also evaluated for AhR activity using a bioassay and examined for developmental abnormalities. TEQs from eggs collected within the Galveston area site ranged from 166 to 452 ng/kg compared to a TEQ of 67 ng/kg for the cormorant egg from one of the reference areas (Telfair Island). TCDD contributed 26 to 51 percent of calculated TEQs with the majority of the remainder being attributable to PCB 126. No deformities or abnormalities in embryos were detected at all sites investigated, suggesting a NOAEL of 452 ng/kg for neotropic cormorant, 376 ng/kg for the black crowned night heron, and 240 ng/kg for the great egret.

- Woodford et al. (1998) examined the survival, growth, and behavior of breeding ospreys exposed to TCDD in Wisconsin. The third eggs of freshly laid clutches were collected over different years (total n=18) from two contaminated sites. Eggs were also collected at two reference sites over three different years (total n=15). Eggs collected from one of the sites had TCDD levels of 29 to 162 ng/kg wet weight while the reference areas had a reduced range from below detection limits to 23.8 ng/kg. Despite the difference in TCDD levels, egg exchange experiments between the affected sites and reference areas showed no significant differences in egg hatching or fledgling rates. A difference (p=0.03) was noted in growth of chicks with the group from the contaminated sites affected. Using the reproductive endpoints of egg hatching and chick fledgling rates suggests a NOAEL of ≥136 ng/kg.
- Custer et al. (2010) investigated the nest and egg success of spotted sandpipers by weekly surveillance of nests on the Hudson River (24 nests) and less frequently in two reference sites (18 nests). Eggs were also collected for chemical analysis (Hudson River, n=13; reference areas, n=5) of PCBs, dioxins, furans and organochlorine pesticides. $TEQ_{DFP,B}$ ranged from 75 to 6,540 ng/kg wet weight on the river and from 8 to 56 ng/kg wet weight for the two reference areas. TEQs at all locations were dominated by PCBs with dioxin and furans contributing fewer than 10 percent of the $TEQ_{DFP,B}$. Results were analyzed using the small sample variant of Akaike Information Criterion to test if nest and egg success was related to TEQ concentration. Models that predicted nest survival and egg success as functions of contaminant levels were poorly distinguished from models that presume no such associations indicating that the contaminant concentrations did not have a sufficient identifiable statistical

relationship with reproductive success. Using the geometric mean $TEQ_{DFP,B}$ concentrations, a NOAEL of 732 ng/kg wet weight is inferred for sandpipers for the Hudson River site.

- Elliott et al. (2001) summarize concentrations of dioxins and furans in great blue heron eggs collected before during and after major changes to the pulp and paper sector in British Columbia, Canada. During the 16-year study, eggs were collected from 21 rookeries either during the egg-laying or incubation period depending on the year. A linear relationship was established between prey fish species and heron egg contamination levels indicating that local dietary uptake was a key exposure route. $TEQ_{DF,B}$ concentrations were elevated through the 1980s at levels sufficient to produce embryotoxicity (reduced chick size, increased brain asymmetry, elevated EROD activities) but decreased markedly in the early 1990s due to changes in bleaching practices of local pulp mills. Although heron $TEQ_{DFP,B}$ concentrations declined after 1990, levels remain near 200 ng/kg ww at one site due to the persistent presence of PCBs. No gross abnormalities or deleterious effects on the number of fledglings were observed at the reference site, Nicomekl River, from which an NOAEC for great blue heron of 207 ng $TEQ_{DFP,B}$/kg ww is estimated.

The field studies described above provides a range of NOAEC values from 136 to 732 ng/kg ww, which cover multiple bird species including the receptors surrogates for the Site. The highest LOAEL value reported in the field (1,700 ng/kg ww) for sandpipers is associated with reduced hatching although there was no effect on nestling survival and growth. In a risk assessment study using multiple lines of evidence (Fredricks et al. 2011), TRVs of 710 and 1,000 ng/kg ww were used as NOAECs for the house wren and eastern bluebird, respectively. The use of these TRVs was supported by field observations of no significant population level effects on reproduction at concentrations below these levels. The geometric mean of NOAECs from three studies on bird receptors present at the site is 411 ng/kg ww (Custer et al. 2010; Elliott et al. 2001; Frank et al. 2001). These field values are consistent with the egg tissue TRVs derived from laboratory yolk injection studies (Table B-6).

2.1.4.2.2 Ingestion Rate TRVs

Research supporting development of NOAELs and LOAELs for birds expressed as an ingested dose of TCDD is rare. For dietary TRVs, only one laboratory study was found in which feeding was the route of administration, conducted on the domestic chicken (Schwetz et al. 1973). USEPA's review of literature from 1981 to 1997 (TN & Associates 2002) also did not identify any studies of acceptable quality reporting ingested doses, and concludes that egg tissue data provide the best means for assessing exposures to birds and evaluating risk (USEPA 2003).

Schwetz et al. (1973) fed 3-day old white leghorn chicks 2,3,7,8-TCDD mixed into food for 21 days using a standardized assay for assessment of chick edema. This study reports a LOAEL for several effects, including chick edema and reduced chick survival of 1 µg/kg diet, with a NOAEL from the same study of 0.1 µg/kg diet. This study was not selected for use in this BERA because some methodological details and test conditions were not included, and the authors report only nominal concentrations in feed, resulting in uncertainty about the actual doses.

Nosek et al. (1992b) conducted experiments with ring-necked pheasant hens, dosing individual hens weekly with intraperitoneal injections of 0, 0.01, 0.1, and 1.0 µg TCDD/kg bw per week. Significant reductions in survivorship reduced egg production and an increase in cumulative egg mortality were observed in hens receiving 1.0 µg/kg per week, or a cumulative dose of 10 µg/kg bw. Affected hens exhibited wasting syndrome. TRVs expressed as a daily dose can be derived from this study. A LOAEL for effects on fertility and hatching success of 1.0 µg/kg bw per week was converted to LOAEL expressed as a daily ingestion rate of 140 ng/kg-day (Nosek et al. 1992b). The dosing regimen was based on orders of magnitude differences and adverse effects were not observed at the next lowest dose, the NOAEL (14 ng/kg-day). Although test subjects were not fed the test chemical, this approach to deriving an oral TRV is appropriate because the exact doses to birds are known, the study is thoroughly reported and robust, and the result is conservative because it assumes 100 percent uptake from an ingested dose, likely an overestimate (Nosek et al. 1992a). It is also in general agreement with Schwetz et al. (1973) despite using different species and routes of administration. Several other reviewers also use results of Nosek et al. (1992b) to derive oral TRVs for use in risk assessment (Sample et al. 1996; Fredricks et al. 2011; Windward 2011a).

2.1.5 Mammals

Toxicity of dioxins and furans in mammals is mediated through AhR, as it is in fish and birds. Exposure of mammals to dioxins and furans is associated with adverse effects on reproduction and development, and the sensitivity of mammals to TCDD toxicity is highly variable. 2,3,7,8-TCDD toxicity in mammals may be characterized by loss of body weight and death. Atrophy of the thymus is consistently a manifestation of 2,3,7,8-TCDD toxicity in mammals, and suppression of thymus-dependent cellular immunity, particularly in young animals, may contribute to their death. Early life stages, including the fetus and newly born pup/kit, appear to be the most sensitive to dioxin toxicity, and maternal exposure can result in increased frequencies of stillbirths. Acute toxicity studies with 2,3,7,8-TCDD have shown marked differences among species; up to a factor of 8,400 between the single oral LD_{50} dose for the guinea pig (the most sensitive mammal) and the hamster (Eisler 1986a).

Unlike fish and birds, only one form of AhR has been shown to exist in mammals. Differences in sensitivity to dioxins and furans may therefore be a function of species-specific toxicokinetic and toxicodynamic factors. The majority of mammalian studies addressing toxicity of dioxins and furans have used common laboratory species (e.g., rat and monkey). Mink have also been the subject of several dioxin and furan exposure studies, and recent literature clearly indicates that 2,3,7,8-TCDF is much less toxic to mink than would be predicted by the mammalian TEF for this congener (Zwiernik et al. 2009). The limited range of mammalian test species argues for caution in extrapolating many of the results to other mammalian wildlife species, which may differ substantially in their life history and ecology from tested animals.

Given the large literature for toxicity of dioxins and furans on rodents and other taxa commonly used in evaluating potential for effects on people, the literature review supporting development of TRVs for mammals is not comprehensive. Studies that are summarized below address survival, growth, or reproduction in mammals.

2.1.5.1 Effects on Growth

In a three-generation reproductive toxicity study of rats, body weight and thymus weight were significantly reduced in third generation female pups in the 0.01 μg TCDD/kg bw-day dietary exposure group (Murray et al. 1979). There were no significant changes in body mass of captive mink exposed to increasing concentrations of dietary TCDF, PeCDF, or a mixture of the two, of up to 9.5 ng TEQ/kg bw-day for up to 6 months (Moore et al. 2009).

Some significant but transient decreases in mink kit mass have been observed, particularly in female kits, at predicted maternal liver concentrations of 36 and 980 ng TCDF/kg liver ww (3.6 and 9.9 ng TEQ/kg liver ww) (Zwiernik et al. 2009). However, in a 3-year field study of trapped mink, adverse effects of a mixture of dioxins and furans on sex ratio, body weight, length, liver weight, and baculum length were not seen at estimated dietary concentrations of 31 ng TEQ/kg ww, a value that was primarily driven by furan content; tissue congener analysis was not reported (Zwiernik et al. 2009). Given Zwiernik et al.'s (2009) finding that TCDF may not be as potent in mink as predicted by the World Health Organization's 2005 TEF for that congener, these studies do not provide a reliable indicator of effects thresholds expressed as TEQ because TCDF is a large component of the exposure in these studies.

2.1.5.2 Reproductive Effects

Several studies document the effects of 2,3,7,8-TCDD on reproduction in mammals. The most applicable to this BERA is a three-generation study with Sprague-Dawley rats exposed continuously to diets with 0.001, 0.01, and 0.1 μg of 2,3,7,8-TCDD/kg bw-day (Murray et al. 1979). No effects were seen at any dose in the first generation rats administered contaminated feed for 90 days, but 0.1 μg /kg bw-day resulted in reduced survival of their neonates. Several reproductive effects were observed in the second generation rats receiving 0.01 μg/kg bw-day, including reductions in rat fertility, decrease in litter size, and significant reductions in the number of pups born alive (Murray et al. 1979). In the same study, pup survival was reduced at the dietary exposure level of 0.01 μg/kg bw-day in second and third-generation rats, but not in the first. Evidence of reproductive toxicity in monkeys is also available: the number of viable offspring born was reduced in groups of female monkeys exposed to 25 ppt TCDD in the diet prior to mating and during gestation and lactation relative to control (Bowman et al. 1989; Schantz et al. 1992).

For mink, older studies generally only report the dose which was lethal to 50 percent of specimens (LD_{50}), or they are feeding studies with fish contaminated with chemicals other than the target dioxin-like compounds, but the data suggest mink are sensitive to toxicity of some dioxin-like compounds. Hochstein et al. (1998) evaluated a range of endpoints in mink fed TCDD at doses ranging from 0.000055 to 3.0 µg /kg-day, resulting in a LOAEL for mortality following 125 day exposure of 0.054 µg /kg-day. A more recent study found no effect on fertility, percentage of kits born alive, or kit survival relative to control in mink administered TCDF at concentrations of 26 or 240 ng TEQ/kg ww diet for three weeks prior to breeding through birth and weaning (Zwiernik et al. 2009). These results illustrate that TCDF is not as toxic in mink as would be expected from the mammalian TEF for that chemical.

Exposure to dioxins and furans during gestation and/or lactation can result in effects in the developing fetus and offspring. However, there is comparatively little empirical evidence for toxicity to the developing female; studies with rats almost exclusively evaluate effects in male pups. Male offspring of exposed mothers exhibit reductions in reproductive organ weights, reduced steroidogenesis, and reduced sperm count (Ohsaka et al. 2002; Hamm et al. 2003; Ikeda et al. 2005; Mutoh 2006). Adverse effects of TCDD on sperm production have been observed in rats (Hamm et al. 2003; Faqi et al. 1998), and effects on sperm production may occur at a lower exposure level (0.25 ng TCDD/g testicular tissue) than the exposure level associated with reduced fertility. Although this is a reproductive effect, reduction in sperm production may have no material effect on wild mammal populations.

Other studies with mammals are available for interpreting exposures to dioxin-like compounds at the Site include Khera and Ruddick (1973) and Kociba et al. (1978). Khera and Ruddick (1973) measured litter size and pup weight in rats exposed to 2,3,7,8-TCDD for 10 days during the period of gestation, reporting a LOAEL of 0.25 µg TEQ/kg-day and a NOAEL of 0.125µg TEQ/kg-day. Kociba et al. (1978) exposed rats to 2,3,7,8-TCDD for 2 years, observing increased mortality in females dosed with 0.1 µg TEQ/kg-day, and no effect on female mortality at 0.01 µg TEQ/kg-day. Because the first of these was a short-term study, and the second did not measure reproductive effects, neither one was used to derive TRVs for this BERA.

In light of uncertainties about the TEFs for mink, because mink are not a receptor at the Site, and because results of the multiple-generation study with rats by Murray et al. (1979) was the only chronic reproductive study, results from Murray et al. (1979) were used to interpret estimated exposures of mammalian receptors for this BERA. The NOAEL of 0.001 µg /kg-day and the associated LOAEL of 0.01 µg/TEQ/kg-day from Murray et al. (1979) were selected as TRVs for mammals (Table B-10).

2.2 Polychlorinated Biphenyls

The toxicity of PCBs to fish, mammals, and birds is relatively well studied. TRVs for exposure of these receptors to PCBs are available as effects levels for three forms: 1) Aroclors (USEPA 2004); 2) total PCBs; and 3) 2,3,7,8-TCDD TEQs. Each of these TRV forms has advantages and disadvantages. Although use of the TEQ approach allows risk analysts to evaluate cumulative exposures and toxicity to multiple compounds, any effects not mediated by the AhR pathway are not accounted for by this method.

Field and laboratory data suggest that many of the toxic effects caused by planar PCBs are mediated subcellularly by AhR, the same receptor responsible for mediating dioxin toxicity. This receptor is involved in the translocation of PCBs into the nucleus and their subsequent binding to AhR (Safe 1991). Because of similar mechanisms of action through binding to AhR, the signs of PCB126 toxicity, for example in lake trout early life stages, are similar to those shown by TCDD, and include yolk-sac edema, multifocal hemorrhages, craniofacial malformation, and mortality (Zabel et al. 1995).

However, although recent work has suggested that while the TCDD-like congeners act by a common mechanism (i.e., AhR), the combined effects of TCDD with the coplanar PCB congeners may not be additive (Walker et al. 1996) because competition for binding sites among the dioxin-like compounds may result in the less potent congeners being the more important driver of response. Despite this uncertainty, the additive model continues to be acceptable for assessing risk because deviation from additivity has been estimated to be within an accepted tenfold range (Walker et al. 1996). TRVs used for interpretation of exposures to dioxin-like PCBs are those discussed and presented in the section on dioxins and furans.

PCBs can produce a variety of responses in organisms and act as neurotoxicants, hepatotoxicants, immunotoxicants, and carcinogens (Safe 1991; Shain et al. 1991; Giesy and Kannan 1998). While sensitivity and responses tend to be species-specific, general responses include lethality, reproductive and/or developmental toxicity, neurotoxicity, hepatic lesions, tumor promotion, suppression of the immune system, and induction of drug-metabolizing enzymes (McFarland and Clarke 1989; Safe and Phil 1990; Eisler and Belisle 1996; Giesy and Kannan 1998). In vertebrates, PCBs induce the cytochrome P450 metabolic enzyme system (Eisler and Belisle 1996). The degree of metabolic breakdown is dependent on the degree of chlorination and the spatial arrangement of chlorine atoms. As the number of chlorine atoms in the PCB molecule increases and the number of unsubstituted adjacent carbon atoms decreases, metabolic transformation decreases. PCB elimination is limited and PCBs bioaccumulate in organisms and biomagnify within food chains.

Of the 209 possible PCB congeners, research has indicated that as much as 75 percent of tissue burdens of PCBs in invertebrates, fish, birds, and mammals consist of only 25 congeners (McFarland and Clarke 1989). These congeners with the greatest likelihood for bioaccumulation and toxicity are the planar non-, ortho-, or mono-ortho-substituted PCBs, which chemically resemble and toxicologically behave similarly to the 2,3,7,8-substituted PCDFs and PCDDs (Walker and Peterson 1991). Specifically, several lines of testing have implicated the planar PCB congeners 77, 81, 126, and 169 as major contributors to the toxicity of PCB mixtures (Ankley et al. 1991).

This section describes TRVs expressed as "total PCBs" based on toxicity studies using Aroclors. Although total PCBs may be calculated as the sum of congeners, the sum of a subset of congeners, or the sum or Aroclors, selection of TRVs expressed as an Aroclors may sometimes be necessary because TRVs expressed as a sum of congeners are not available. In these cases, the exposures based on sum of Aroclors or sum of congeners is compared to a TRV expressed as an Aroclor, a conservative approach.

TRVs expressed as total PCBs are included for use in the BERA. Risks evaluated using total PCBs as the exposure metric are considered separately (not additively or cumulatively) from risks associated with the dioxin-like toxicity of PCBs. Site-specific data allow for estimates of exposure to total PCBs by most receptors, and include: all 209 congeners in tissue samples collected for the Site; Aroclors and dioxin-like congeners at 12 soil stations, and Aroclors in surface and subsurface soils for the southern impoundment area; surface sediment data for Aroclors and dioxin-like congeners collected for the remedial investigation and a few samples at one location (under the I-10 bridge) collected by TCEQ; and surface water data for all 209 congeners in a few samples collected by TCEQ in one location under the I-10 bridge.

There is uncertainty associated with the use of Aroclor 1254 toxicity information in combination with total PCBs as the exposure metric. The mixture of PCB congeners in sediments and tissue at the Site may not reflect the same congener composition as Aroclor 1254. Nevertheless, the assessment approach should be protective because Aroclor 1254 is expected to be among the Aroclors most toxic to birds and mammals based on extrapolation of comparative studies of Aroclors in aquatic organisms (Nebeker and Puglisi 1974; Mayer et al. 1977; Johnson and Finley 1980). Moreover, dechlorination of PCBs by natural processes at the Site would likely lead to mixtures with toxicity less than or equal to Aroclor 1254, because Aroclor 1254 is a mixture of highly chlorinated PCBs, which generally have relatively high toxicity. PCB toxicity to birds and mammals is addressed in the BERA with approaches based on both total PCB exposures and $TEQ_{P,B}$ and $TEQ_{P,M}$ exposures, respectively. The finding of negligible risk for the Site based on $TEQ_{P,F}$, $TEQ_{P,B}$, and $TEQ_{P,M}$ supports the overall conclusion of negligible risk to fish, birds, and mammals from PCBs, improving confidence in similar conclusions for analyses based on total PCBs.

2.2.1 Fish

The effects of PCBs on Great Lakes fish and wildlife have been extensively studied. PCB-induced reproductive impairment has been demonstrated for several fish species (Ankley et al. 1991; Mac 1988; Walker et al. 1991a; Walker et al. 1991b; Walker and Peterson 1991; Williams and Giesy 1992). Generally, the most sensitive endpoints for effects of PCBs in fish are early life-stage survival and recruitment where exposure has resulted from transfer of PCBs from maternal tissue to eggs (Eisler and Belisle 1996; Walker et al. 1996). Whole-body concentrations of PCBs in adult fish that are commonly found in the environment do not generally result in death (Eisler and Belisle 1996). This is consistent with numerous field studies evaluating PCB fish tissue concentrations and adverse effects summarized by Niimi (1996). Based on several field studies, lethal body burden concentrations have been estimated at greater than 100 mg/kg for young fish and greater than 250 mg/kg for older fish (Niimi 1996).

Numerous studies report TRVs as residues in tissue of fish administered PCBs through water only, food only, or water and food combined. The tissue-based NOAEL and LOAEL for this risk assessment were developed primarily from the literature. The derivation of TRVs focused on the measurement endpoints related to survival, growth, and reproduction and, at USEPA's request, data for freshwater fish species were not included. The methodology used to combine data when deriving the tissue-based TRVs was analogous to that used to derive USEPA EcoSSLs for soils.

2.2.1.1 Derivation of NOAEL for Total PCBs in Whole Fish

There were several studies reported in the literature or used by state or federal agency ecological risk assessments that reported NOAEL values for fish expressed as whole-body concentrations that were included in calculation of the NOAEL for this BERA.

- Hansen et al. (1973) exposed female sheepshead minnow (*Cyprinodon variegatus*) to Aroclor 1254 in water using a flow-through system. Fish were exposed for 28 days to control water or five nominal concentrations of Aroclor1254 in water (0.1, 0.32, 1.0, 3.2, and 10 μg/L). All fish survived and egg production was induced. The eggs were fertilized and placed in PCB-free flowing seawater and observed for mortality. Survival of fry to 1 week of age was 77 percent for eggs from adults from the 0.32 μg/L concentration in water treatment (average 9.3 mg/kg in tissue of females), as compared to 95 percent survival of fry from control adults and 97 percent survival of fry from adults from the NOAEL treatment (0.1 μg/L; average 1.9 mg/kg in tissue of females). This study was used to derive the tissue-based NOAEL and LOAEL of

1.9 mg/kg and 9.3 mg/kg, respectively, for both the Hudson River Revised BERA (USEPA 2000b) and Onondaga Lake BERA (NYSDEC 2002).[4]

- Bengtsson (1980) exposed adults of the common minnow (*Phoxinus phoxinus*; also called the Eurasian minnow) to the PCB mixture Clophen A50. Clophen A50 contains 50 percent chlorine by weight and has similar physicochemical parameters to Aroclor 1248. Fish were exposed to a control diet, or a diet fortified at three PCB levels (20, 200, and 2,000 mg/kg) for 40 days, and then monitored for a total of 300 days. Fish were subsampled at several times during this period and their whole-body PCB concentrations were quantified. Growth, reproduction, and behavioral effects (i.e., swimming) were monitored during this period. There was no apparent impact on hatchability of the ova from exposed adults for average whole-body tissue concentrations up to 15 mg/kg ww (this corresponded to the 200 mg/kg diet exposed fish). The value of 15 mg/kg ww represents the NOAEL.
- Westin et al. (1983) fed striped bass (*Morone saxatilis*) larvae PCB-contaminated brine shrimp (*Artemia* spp.) from yolk sac absorption to either 10 or 20 days. Larvae were left to deplete the yolk sac for the first 10 days, and then fed the PCB-contaminated shrimp for 20 days. Larvae were subsampled at test initiation, at post-yolk sac (day 10), after 10 days of feeding (day 20), and after 20 days of feeding (day 30), and analyzed for PCBs. Survival and growth were monitored during the study period. There was no apparent impact on survival or growth in the treatment, and PCB concentrations in tissue were found to decline over the monitoring interval, which was attributed to growth dilution. The highest post-yolk sac whole-body tissue concentration was in the larvae fed contaminated shrimp for 10 days, and was 4.4 mg/kg ww. This concentration is included as a NOAEL, as required by USEPA in comments on the draft of this report (Appendix F). Uncertainty associated with this result is due to the lack of any observed effect (i.e., the NOAEL is unbounded) and the fact that the study period encompassed a period of rapid larval growth and consequent dilution of the dose into the larval tissues. These uncertainties result in a NOAEL that is lower than it would be if the test organisms were not undergoing a high rate of growth at the time of dosing, and that is not clearly representative of any particular life stage.

[4] This paper was cited as "Hansen et al. (1974)" in USEPA (2000) and NYSDEC (2002), but the correct publication year was 1973.

2.2.1.2 Derivation of LOAEL for Total PCBs in Whole Fish

Two of the studies that are discussed above (Bengtsson 1980 and Hansen et al. 1973) also reported LOAEL values. In the Bengtsson (1980) study, the individuals with the highest exposure (2,000 mg/kg in their diet) had an average whole-body total PCB concentrations of 170 mg/kg. Eggs of these fish exhibited decreased hatchability, so 170 mg/kg is the LOAEL derived from Bengtsson (1980). The Hansen et al. (1973) study was used in the draft Hudson River BERA (USEPA 1999b) to derive the TRV, but was not used for the Hudson River Revised BERA (USEPA 2000b). The Hansen study LOAEL was 9.3 mg/kg.

An additional study by Orn et al. (1998) was included in calculation of the LOAEL, as required by USEPA in comments on the draft (Appendix F). Orn et al. (1998) purchased adult zebrafish (*Danio rerio*), and after 4 weeks of acclimatization, exposed the fish to a mixture of 20 selected PCB congeners at three dose levels in feed for 13 weeks. The reproduction study was initiated following 9 weeks of exposure. Survival, growth, and reproduction were monitored during the study, along with various histopathological endpoints (e.g., the liver somatic index) that required removal of the subject organs from the dosed females. A reduction in the number of eggs per female and reduced larval survival were observed in the high dose group, resulting in a LOAEL of 2.7 mg/kg, the concentration in the females after livers and ovaries were removed. Potential confounding factors in this study include uncertainties about the representativeness of the 20 selected PCB congeners of mixtures to which fish may be exposed, and the fact that ovaries and liver were removed prior to tissue PCB analysis. As a result of the organ removals, it is likely that the LOAEL is a substantial underestimate because PCBs accumulate in ovary and liver tissue of fish.

The recommended NOAEL and LOAEL values are 5.0 and 16 mg/kg ww in whole body fish tissue, respectively (Table B-11). These TRVs are the geometric mean TRVs derived from the studies accepted for TRV development as described above and summarized in Table B-12.

2.2.2 Reptiles

The possibility that exposure of reptiles to PCBs and related elevated concentrations in turtle eggs could result in deformities in developing turtles is discussed in Section 1.3 and 2.1.3.

Exposure-response relationships to describe effects of PCB exposure on turtles and other reptiles have not been developed, and acceptable TRVs for interpreting exposures to reptiles on the Site are not available.

2.2.3 Birds

Effects of ingested PCBs on birds include disruption of normal patterns of growth, reproduction, metabolism, and behavior. PCB-induced reproductive impairment has been demonstrated for a number of insectivorous and piscivorous birds (Gilbertson et al. 1991; Kubiak et al. 1989; Tillitt et al. 1992) and is generally the most sensitive endpoint, with effects on fertility, egg production, and hatching success (Eisler 1986b). Reduced survival of offspring and growth effects in offspring through the F2 generation have also been demonstrated (American kestrel studies by Fernie et al. 2003a,b,c).

Chickens and other gallinaceous birds (e.g., pheasant) are among the most sensitive species tested for effects of PCBs and dioxins. Among studies with non-gallinaceous birds, the passerine northern bobwhite (*Colinus virginianus*) appears to be less sensitive to effects of ingested Aroclors (Heath et al. 1969; Scott 1977). Bird embryos are the most sensitive life stage for assessing the effects of contaminants (Elliott et al. 1996; Kubiak and Best 1991).

Avian TRVs for dietary exposure were developed using Aroclor 1254 or "environmental PCBs" that might be representative of the exposure pathways that could occur at the Site. Only those studies that were conducted over at least a 2-month period were included in this assessment. This was done because there are large number of LD_{50} toxicological studies or short-term (e.g., single dose) studies that are not relevant to environmental exposures. The studies included in the dietary-TRV derivation are summarized briefly below.

Peakall (1971) exposed ring doves (*Streptopelia risoria*) to a diet containing 10 ppm of Aroclor 1254 for 6 months and evaluated whether there was any impact on eggshell thickness (based on washed eggshell weights) relative to a control diet. The eggshell weights of exposed and control birds were comparable. The 10 ppm represents a NOAEL. The author did not include the body weight or ingestion rates of the test organisms. Using the average body weight (0.155 kg) and ingestion rate (0.017 kg/day) reported by Sample et al.

(1996), this equates to a NOAEL of 1.1 mg/kg-day. A LOAEL could not be calculated from this study.

Dahlgren et al. (1972) exposed ring-necked pheasants (*Phasianus colchicus*) for 17 weeks via oral gavage and monitored reproduction. Two dose levels were used (12.5 and 50 mg/kg of Aroclor 1254). USEPA (2000b) considered the lower dose to be a NOAEL and the upper dose the LOAEL. Adjusting to a daily dose using the average body weights of the test organisms, yielded a NOAEL and LOAEL of 1.8 and 7.1 mg/kg-day, respectively.

Heath et al. (1972) evaluated the toxicity of Aroclor 1254 in the northern bobwhite (*Colinus virginianus*) and Mallard duck (*Anas platyrhynchos*). Over a 2-year period, birds were fed diets containing either 25 or 50 ppm of Aroclor 1254 and were evaluated for egg production, egg hatchability, and survival of chicks. The NOAEL for the northern bobwhite was 50 ppm in the diet, while the NOAEL for the duck was 25 ppm (and the LOAEL was 50 ppm). Adjusting to a daily dose using the average body weights and ingestion rates of the test organisms resulted in NOAEL values of 4.7 and 7 mg/kg-day for the northern bobwhite and duck, respectively, and a LOAEL of 14 mg/kg-day for the duck.

Platonow and Reinhart (1973) evaluated the toxicity of Aroclor 1254 in the chicken (*Gallus domesticus*). Birds were fed diets containing either 5 or 50 ppm of Aroclor 1254 for 39 weeks. The 50 ppm dose significantly reduced production of eggs and hatchability, and was replaced with the control ration after 14 weeks. The 5 ppm level of PCB reduced egg production but not hatchability of fertile eggs. Fertility of eggs in the 5 ppm group also declined after 14 weeks of exposure, but the authors reported that this was not related to PCB exposures. Therefore, the 5 ppm level represented a LOAEL. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used to develop TRVs. The 5 ppm dose level resulted in a LOAEL of 0.35 mg/kg-day. A NOAEL could not be calculated from this study.

Peakall and Peakall (1973) evaluated the second-generation ring doves from their prior study that exposed the first-generation group to a diet containing 10 ppm of Aroclor 1254 (Peakall 1971). The second generation doves were inconsistent in incubation of their eggs resulting in reduced hatchability. The dietary value of 10 ppm was considered a LOAEL, which yielded a LOAEL of 1.1 mg/kg-day using the same assumptions for ingestion rate and body weight as used for evaluating the Peakall (1971) study. A NOAEL could not be calculated from this study.

Cecil et al. (1974) evaluated the toxicity of Aroclors 1242, 1248, and 1254 in the chicken (*G. domesticus*). For 9 weeks, birds were fed diets containing either 2 or 20 ppm of the mixed PCBs. Hatchability declined 2 weeks after hens were given the upper dose but there was no effect at the lower dose. Therefore, the 2 ppm dose represented the NOAEL and the 20 ppm dose represented the LOAEL. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used to derive TRVs. The calculated NOAEL and LOAEL were 0.14 and 1.4 mg/kg-day, respectively.

Lillie et al. (1974) evaluated the toxicity of Aroclor 1254 in the chicken (*G. domesticus*). For 9 weeks birds were fed diets containing either 2 or 20 ppm of the Aroclor 1254. There were no effects to adult body weight gain, survival, egg weight, eggshell thickness, or fertility at either dose level. Egg production was significantly reduced relative to control at the 20 ppm dose level. Feed consumption of adults was also depressed at this dose level, which may have contributed to the reduced egg production. Based on these results, the 2 ppm dietary level represents the NOAEL while the 20 ppm dose level represents the LOAEL. The authors did not report body weights for the 9-week exposure period, but did report the initial average body weight (1.953 kg) which was used for the TRV calculation. The authors reported a food consumption rate of 118.5 to 124.3 g/day (mean: 121.4 g/day) for the two dose levels for the 9-week period. The mean ingestion rate was used to calculate the TRVs. The calculated NOAEL and LOAEL were 0.124 and 1.24 mg/kg-day, respectively.

Lillie et al. (1975) evaluated the toxicity of Aroclor 1254 (as well as Aroclors 1016, 1232, 1242, and 1248) in the chicken (*G. domesticus*). Birds were fed diets containing 5, 10, or 20 ppm of the Aroclor 1254 for 8 weeks. There were no adverse effects on egg production, egg weight, eggshell thickness, feed consumption, adult body weight changes, survival, or fertility during this exposure period. Based on these results, the 20 ppm dietary level represents the NOAEL. Using the average body weights and ingestion rates reported from their prior study (Lillie et al. 1974), the NOAEL was 1.24 mg/kg-day.

Riseborough and Anderson (1975) exposed mallard ducks (*Anas platyrhynchos*) to a diet containing 40 ppm of Aroclor 1254 for approximately 4 months and monitored egg production, eggshell thickness, and related endpoints. There were no differences between ducks fed a control diet or the 40 ppm diet on any of the measured endpoints. Based on these results, the 40 ppm dietary level represents a NOAEL. The authors did not report average body weights or ingestion rates. Therefore, the values reported by Sample et al. (1996) for these terms were used to calculate the NOAEL of 4.0 mg/kg-day.

Kosutsky et al. (1979) exposed chickens to a diet containing 5 ppm of the PCB mixture Delor 105, which is 54 percent chlorine by weight (similar to Aroclor 1254). Birds were fed this diet for 6 weeks followed by 3 weeks of control diet. There were no differences relative to controls for egg production, egg weight or eggshell strength and weight. The authors did not report body weights or ingestion rates, so values reported in USEPA (1993) and Sample et al. (1996) were used for these terms to estimate a TRV. The calculated NOAEL was 0.35 mg/kg-day.

Roberts et al. (1978) reported a study where ring-necked pheasants (*P. colchicus*) were exposed to Aroclor 1254 at a dietary concentration of 50 ppm. In its review of this study, USEPA (2000b) reported that there was a reduction in female fertility at this dose level, and a LOAEL of 2.9 mg/kg-day was calculated.

Custer and Heinz (1980) exposed ducks to diets containing 25 ppm of Aroclor 1254 for 1 month. There was no apparent effect on reproductive success during this period. Although a NOAEL (7.0 mg/kg-day) could be calculated from this study, it did not meet the minimum of 2 months exposure used to derive TRVs for this project.

Summer et al. (1996) exposed white leghorn chickens *(G. domesticus*) to diets containing carp collected from Saginaw Bay, Lake Huron, Michigan. The diets contained 0, 3.5, or 34 carp, which yielded total PCB concentrations in the diets of 0.3, 0.8, and 6.6 mg/kg (respectively). The chickens were fed for an 8-week period, which overlapped egg-laying. Food consumption rates were similar across the dose groups and exposure periods. The mean body weights decreased with increasing dose and exposure periods, although the authors did not evaluate whether these were statistically significant. On average, the daily egg production and egg weights were greater with the diets containing carp relative to control. These results would suggest a potential NOAEL for growth only at the intermediate dose level. Based on the average body weight for the intermediate dose group (1.593 kg), and their average food consumption rate (91.19 g/day), the calculated NOAEL is 0.046 mg/kg-day. This value was excluded from calculation of the geometric mean TRV because of the potential influence of other chemicals.

Custer et al. (1998) evaluated the reproductive success of tree swallows (*Tachycineta bicolor*) exposed to environmental PCBs in the Fox River and Green Bay systems. Prey items (emergent insects were collected for chemical analysis). The authors reported that there were no effects on clutch size or egg hatchability in adults that had consumed diets containing up to 0.61 mg/kg of total PCBs. Based on this information, USEPA (2000b) reported a NOAEL of 0.55 mg/kg-day. This value was excluded from calculation of the geometric mean TRV because of the potential influence of other chemicals.

The geometric means of the NOAEL and LOAEL from these studies (excluding Custer and Heinz [1980], Summer et al. [1996], and Custer et al. [1998]) are 2 and 3 mg/kg-day, respectively (Table B-11).

2.2.4 Mammals

TRVs for total PCBs were derived for mink and other mammals based primarily on reproductive toxicity studies in the literature. Toxic responses of mammals to PCB exposure are highly species-specific, and younger mammals appear to be more susceptible to PCB effects than adults (Eisler 1986b). PCB-induced reproductive impairment has been demonstrated for mink (Bleavins et al. 1980; Heaton et al. 1995a,b; Tillitt et al. 1996; Wren 1991) and other mammals including mice, rats, rabbits, swine, and rhesus monkeys (Villeneuve et al. 1971; Golub et al. 1991). Mink are generally regarded as the most sensitive mammal to ingested PCBs. Caution is needed when interpreting studies because study designs differ widely. In particular, studies in which mink are fed contaminated fish collected from the field are confounded by the possibility that other chemicals were present in the fish used to dose the test animals.

2.2.4.1 PCB TRVs for Mammals Other Than Mink

Review of the ECOTOX database showed that there is a greater frequency of toxicity data reported for mice than rats, so the TRV review for PCBs focused on mice. The key studies were compiled and are briefly summarized below. Geometric means of the NOAELs and LOAELs were calculated from results of these studies, and results were used as the NOAEL and LOAEL for mammals used in this BERA (Table B-13).

Linzey (1987) evaluated reproductive success in wild caught and laboratory-reared white-footed mice (*Peromyscus leucopus*) exposed to 10 ppm of Aroclor 1254 in their diets. There was a statistically significant reduction in the number of surviving offspring per litter in the PCB-exposed mice wild caught, but no effect on other reproductive parameters (e.g., litter size at birth). Based on this information, the 10 ppm dose level represents a LOAEL. Using the average body weights (23.2 g) and average food consumption rate (0.127 g/g bw-day) reported by the author, the calculated LOAEL is 1.27 mg/kg-day.

Linzey (1988) evaluated the survival and growth of the second generation of mice from Linzey (1987) study. At the same dose level of 10 ppm of Aroclor 1254, the second generation of PCBs-treated offspring exhibited poor reproductive success relative to controls, and grew at a slower rate compared to controls. The same LOAEL calculated from Linzey (1987) is applicable to this current study.

Simmons and McKee (1992) fed white-footed mice (*P. leucopus*) diets containing Aroclor 1254 at four dietary levels (2.5, 25, 50, and 100 ppm) for 21 days and monitored survival. There was no effect of PCB exposure at the 2.5 ppm diet concentration and a slight effect at 25 ppm. The latter represents the LOAEL and the 2.5 ppm level a NOAEL. Based on the average body weight and ingestion rate, this yields NOAEL and LOAEL values of 0.36 and 3.6 mg/kg-day, respectively. The NOAEL corresponds to the "TRV-low" recommended by USEPA Region 9 (USEPA 2002).

McCoy et al. (1995) exposed three generations of old-field mice (*Permyscus polionotus*) to a diet containing 5 ppm of Aroclor 1254 for 12 months and monitored reproduction. Dietary exposure reduced the number of litters, offspring weights, and offspring survival. Sample et al. (1996) concluded that this dietary level represented a LOAEL, and based on literature values for body weights and ingestion rates (this information was not provided by the authors), derived a LOAEL of 0.68 mg/kg bw-day.

Voltura and French (2007) fed breeding female white-footed mice (*P. leucopus*) for 4 months on diets containing a mixture of Aroclors 1242 and 1254 (ratio of 2:1) at dietary levels of 10 and 25 ppm and monitored reproductive success. There was no effect of PCB exposure on litter size at birth or weaning, although there was a statistically significant reduction in reproductive success in female mice that were fed the 25 ppm diet. Based on this information, the latter represents the LOAEL and the 10 ppm level a NOAEL. The authors calculated daily ingestion rates of 2.64 mg/kg-day for the 10 ppm diet and 6.19 mg/kg-day for the 25 ppm diet, which represents the NOAEL and LOAEL, respectively.

The geometric means of the mouse NOAELs and LOAELs from these studies are 0.98 and <2 mg/kg-day, respectively. These values are used in the BERA risk calculations for assessing PCB exposure to marsh rice rat and raccoon. Results of studies with mink were not used in calculations of hazard quotients, but provide information relevant to the uncertainty analysis and are discussed below.

2.2.4.2 PCB TRVs for Mink

Mink are not a receptor at this Site and are unusually sensitive to PCB toxicity, so the TRVs used to evaluate risk to mammals did not include TRVs for mink. The geometric mean of NOAELs reported by several studies is 0.2 mg/kg-day, and is an unbounded NOAEL value. The geometric mean of LOAELs reported by several studies is also 0.2 mg/kg-day (Table B-11). These values are not used for the BERA risk calculations for assessing PCB exposure to marsh rice rat and raccoon. Because mink may be more sensitive than rice rat and raccoon, risks to these receptors were calculated using the PCB TRVs for mammals other than mink that are discussed above.

2.3 Bis(2-ethylhexyl)phthalate

BEHP was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

2.3.1 Benthic Invertebrates

No ER-L/ER-M values or AWQC for BEHP were available for use in the evaluation of risks to benthic invertebrates. Ho et al. (1997) measured an LC_{50} of > 1,000 μg/L for BEHP in opossum shrimp and amphipods exposed for 4 days (USEPA 2012). These were the lowest LC_{50}s in the ECOTOX database. Two other studies measured higher LC_{50}s in copepods exposed for 4 days (1 x 10^6 μg/L and 3 x 10^6 μg/L). The LC_{50} from Ho et al. (1997) was divided by 10 to estimate an NOAEC of 100 μg/L in surface water. There were several NOAECs for crustaceans and one for a polychaete that were higher than 100 μg/L. This NOAEC was used as the TRV for BEHP in benthic invertebrates (Table B-14).

2.3.2 Fish

No AWQC for BEHP were available for use in the evaluation of risks to fish. ECOTOX lists a study by Heitmuller et al. (1981) that observed an LC_{50} of 550,000 μg/L in sheepshead minnows exposed for 4 days (USEPA 2012). This value was divided by an uncertainty factor of 10 to derive an NOAEC of 55,000 μg/L, which was used as the TRV for BEHP in fish (Table B-15). There were several NOAECs for fish listed by ECOTOX that were higher than this value, but there were no studies with longer exposure durations.

2.3.3 Birds

No EcoSSL is available for BEHP and it is not addressed by Sample et al. (1996). A literature review identified only one study relevant to avian toxicity. O'Shea and Stafford (1980) studied feeding rates, weight gain, and bioaccumulation of BEHP in starlings and found that wild starlings fed diets up to 260 mg/kg BEHP for 30 days did not accumulate BEHP and had higher body weights than control birds. This dietary concentration was converted to a NOAEL dose of 74.88 mg/kg-day assuming a consumption rate of 21.6 g food/day (measured in the study) and a body weight of 75 g (based on Cuthill et al. 1999). This NOAEL was selected for use in this BERA (Table B-16). No LOAEL was identified in this study.

2.3.4 Mammals

Testicular toxicity is considered a critical effect for BEHP in mammals (ATSDR 2002). ATSDR (2002) reviewed eight studies of reproductive effects in rodents fed BEHP in food for

1 to 2 years. The bounded NOAEL for reproductive effects was 5.8 mg/kg-day based on bilateral testicular aspermatogenesis in rats following 104 days of exposure; the associated LOAEL was 29 mg/kg-day (David et al. 2000, as cited in ASTDR 2002). NOAELs associated with survival and growth endpoints were higher. The NOAEL of 5.8 mg/kg-day and LOAEL of 29 mg/kg-day were selected for this BERA (Table B-10).

2.4 Carbazole as a $COPC_E$ for Benthic Invertebrates

Carbazole was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). Carbazoles are natural products of some marine organisms (Pindur and Lemster 2001) and, as such, may represent a low level of risk to marine organisms at concentrations commonly present in the marine environment. No ER-L/ER-M values, no AWQC, and no ECOTOX records were available to describe the toxicity of carbazole to marine invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

2.5 Phenol as a $COPC_E$ for Benthic Invertebrates

Phenol was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values or AWQC for phenol were available for use in the evaluation of risks to benthic invertebrates. The ECOTOX database lists a study by Kim and Chin (1995) reporting an LC_{50} of 260 µg/L for a mysid shrimp (*Archaeomysis kokuboi*) exposed for 4 days (USEPA 2012). This was the lowest LC_{50} value in the ECOTOX database for a marine invertebrate. The other LC_{50} values with exposure durations ranging up to 21 days were higher, ranging from 5,800 to 1.05×10^8 µg/L. The LC_{50} value from Kim and Chin (1995) was divided by 10 to estimate an NOAEC of 26 µg/L in surface water, which was used as the TRV for phenol for benthic invertebrates (Table B-14).

3 METALS

Metals considered $COPC_{ES}$ for one or more ecological receptors are aluminum, barium, cadmium, cobalt, copper, lead, manganese, mercury, nickel, thallium, vanadium, and zinc (Table B-1). Each of these $COPC_{ES}$ is discussed below.

3.1 Aluminum as a $COPC_E$ for Benthic Invertebrates

Aluminum was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values, no AWQC, and no ECOTOX records were available for aluminum for use in the evaluation of risks to benthic invertebrates. Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96 hours and measured an LC_{50} value of 10,000 μg/L for aluminum. This LC_{50} was divided by 10 to estimate a NOAEC of 1,000 μg/L, which was used as the TRV for aluminum in benthic invertebrates (Table B-14).

3.2 Barium as a $COPC_E$ for Benthic Invertebrates

Barium was selected as a $COPC_E$ for benthic invertebrates (Table B-1). Barium is a naturally occurring metal used in the manufacture of ceramics, pyrotechnics, paints, enamels, and television tubes, and can be released to the environment through related industrial processes and through coal and oil combustion. Barium is more soluble in sandy soils with low pH and low organic carbon content. In biota, the properties of barium allow it to replace calcium, particularly in the release of neurotransmitters and adrenal catecholamines (USEPA 2005a).

No ER-L/ER-M values, no AWQC, and no ECOTOX records were available for barium for use in the evaluation of risks to benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

3.3 Cadmium

Cadmium can be absorbed by mammals via respiration and ingestion; absorption of ingested cadmium is controlled by several factors including the age of the organism, the valence state or form ingested, and the presence of foods rich in protein or calcium (USEPA 2005b). Metal-binding, proteinaceous metallothioneins appear to protect vertebrates from deleterious effects of high metal body burdens (Eisler 1985). Cadmium bioconcentrates, primarily in the liver and kidney (USEPA 1999a). Cadmium accumulated from water is slowly excreted, while cadmium accumulated from food is eliminated more rapidly.

Cadmium was selected as a $COPC_E$ for fish, reptiles, birds, and mammals (Table B-1).

3.3.1 Fish

Windward (2011b) performed a review of toxicity studies in fish that were fed cadmium for various periods. The study with the lowest bounded NOAEC (in food of fish) of 68 mg/kg ww was converted to 9.0 mg/kg dw based on Windward's reported moisture content estimate of 86.7 percent for the fish diet. This study also reported the lowest bounded LOAEC of 106 mg/kg ww, or 14.1 mg/kg dw (Hatakayama and Yasuo 1987, as cited in Windward 2011b) for a reduction in the number of fry produced by guppies exposed to cadmium in food for a period of 7 months. Several other unbounded NOAECs were identified by Windward (2011b). The NOAEC of 9.0 mg/kg dw in food was used as the TRV for cadmium in fish (Table B-15).

3.3.2 Birds

Birds are comparatively resistant to the toxicity of cadmium, and mallards and chickens have been reported to tolerate 200 mg/kg of cadmium in diets for protracted periods. When present at sufficiently high doses, sublethal effects of cadmium in birds are similar to those in other animals and include growth retardation, anemia, and testicular damage. To develop an EcoSSL for cadmium, USEPA reviewed 35 papers that evaluated toxicity of cadmium to birds (USEPA 2005b); these included 49 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA calculated a TRV of 1.47 mg/kg-day, based on the geometric mean of the NOAELs for growth and reproduction (USEPA 2005b). This value was selected as the NOAEL for this BERA. The NOAEL is lower than the lowest bounded LOAEL of 2.37 mg/kg-day for reproductive effects in a 12-month dietary study of chicken (USEPA 2005b), which was selected as the LOAEL for this BERA (Table B-16).

3.3.3 Mammals

Mammals are relatively resistant to the toxicity of cadmium. Absorption and retention of cadmium decrease with prolonged exposure (Eisler 1985). Cadmium absorption through ingestion is inversely proportional to intake of other metals, especially iron and calcium.

USEPA identified 145 acceptable papers containing data for cadmium toxicity to mammals (USEPA 2005b). Within these papers were 141 NOAEL and/or LOAEL results related to survival, growth, or reproduction. The geometric mean of 38 bounded NOAELs for survival, growth, or reproductive endpoints is 2 mg/kg-day. The geometric mean of the associated LOAELs is 10 mg/kg-day. These values were used as the TRVs for cadmium in mammals (Table B-10).

3.4 Chromium

Chromium was selected as a $COPC_E$ for reptiles, birds, and mammals (Table B-1).

3.4.1 Birds

USEPA (2008) reviewed the literature on avian toxicity of trivalent and hexavalent chromium and identified 13 studies with relevant data. Insufficient study results were available to derive a TRV for hexavalent chromium, but there were 18 results related to survival, growth, or reproduction in birds exposed to trivalent chromium. USEPA calculated a TRV of 2.66 mg/kg-day based on the geometric mean of the NOAELs for growth and reproduction (USEPA 2008). The lowest bounded LOAEL is 2.78 mg/kg-day, from a 180 to 190 day feeding study that found reproductive effects in black ducks (Table B-16).

3.4.2 Mammals

USEPA's (2008) review of the chromium toxicity literature identified 20 studies with data for mammalian test species; these included 16 NOAEL and/or LOAEL results for survival, growth, or reproductive endpoints in mammals exposed to trivalent chromium and 46 results for mammals exposed to hexavalent chromium. The TRVs for trivalent and hexavalent chromium, each based on the geometric mean of the NOAELs for growth and reproduction, are 2.40 and 9.24 mg/kg-day, respectively. The more conservative value of 2.40 mg/kg-day for trivalent chromium was selected as the NOAEL for this BERA. There are no bounded LOAELs for trivalent chromium studies. The lowest unbounded LOAEL for trivalent chromium is 2.82 mg/kg-day, based on a 50-day dietary study that observed mortality in rats. The lowest bounded LOAEL for hexavalent chromium is higher. The LOAEL of 2.82 mg/kg-day for trivalent chromium was selected for this BERA (Table B-10).

3.5 Cobalt as a $COPC_E$ for Benthic Invertebrates

Cobalt was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values or AWQC are available for cobalt for use in the evaluation of risks to benthic invertebrates. The only ECOTOX record related to cobalt's effects on survival, growth, or reproduction was a report of general effects on growth at 10 µg/L in a 14-day study of Pacific oysters (Watling 1983, as cited in USEPA 2012). The result was not indicated as statistically significant. This result could not be used as the basis for a TRV for cobalt in benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles.

Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96-hour and measured an LC_{50} value of 4,500 µg/L for cobalt. The LC_{50} was divided by 10 to estimate a NOAEC of 450 µg/L, which was used as the TRV for cobalt in benthic invertebrates (Table B-14).

3.6 Copper

Copper was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1). Copper occurs naturally in many animals and plants and is an essential micronutrient that animals incorporate into several enzymes. Adverse effects in vertebrates exposed to copper include hematological, hepatic, developmental, immunological, and renal impairment. Copper exerts toxic effects by binding to DNA or by generating free radicals (USEPA 1999a). Aqueous copper speciation and toxicity depend on the ionic strength of the water. Primarily it is the dissolved cupric ion (Cu^{2+}) and possibly hydroxyl complexes that are toxic to aquatic biota; copper complexes consisting of carbonates, phosphates, nitrates, ammonia, and sulfates are weakly toxic or nontoxic (USEPA 2000a). In hard waters, 43 to

88 percent of the copper is associated with suspended solids and not available to biota (Eisler 1998).

3.6.1 Benthic Invertebrates

Many aquatic species are sensitive to dissolved concentrations of copper in the range of 1 to 20 µg/L (USEPA 2000a). In aquatic invertebrates, copper causes gill damage at high concentrations, and in fishes it interferes with osmoregulation (Eisler 1998). The AWQC for copper are 3.1 µg/L (CCC) and 4.8 µg/L (CMC) for chronic and acute exposure, respectively (USEPA 2009). For copper in sediment, the ER-L (i.e., the concentration below which adverse effects rarely occur) is 34 mg/kg dw, and the ER-M (i.e., the concentration above which adverse effects are considered probable) is 270 mg/kg dw (NOAA 1999). The ER-L of 34 mg/kg and ER-M of 270 mg/kg in sediment were used as TRVs for copper in benthic invertebrates.

3.6.2 Fish

Windward (2011b) reviewed 15 toxicity studies in fish that were fed copper for various periods. All of the studies reported on growth effects; studies reporting reproductive effects were not found. The study providing the lowest bounded NOAEC and LOAEC could not be confirmed by other investigators and was not considered typical of toxicity levels in other fish species. As a result, Windward (2011b) selected the next-lowest results of a 60-day feeding study with juvenile rockfish reporting an NOAEC of 50 mg/kg and an LOAEC of 100 mg/kg dw for growth. These were used as the TRVs for copper expressed as a concentration in fish food (Table B-15).

3.6.3 Birds

Experiments with domestic poultry show that copper accumulates in livers of mallard ducklings at dietary concentrations as low as 15 mg/kg dw ration. Mehring et al. (1960) reported a NOAEL of 570 mg/kg copper and a LOAEL of 749 mg/kg dw for dietary copper exposure of chicks over a period of 10 weeks. Using standard assumptions regarding body weight (0.534 kg) and food consumption (0.044 kg/day), Sample et al. (1996) derived a NOAEL of 47 mg/kg-day and a LOAEL of 62 mg/kg-day.

USEPA (2007a) identified 107 studies with data for avian test species; these contained 205 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007a) identified a TRV of 4.05 mg/kg-day, based on the highest bounded NOAEL below the lowest bounded LOAEL, which came from an 84-day feeding study of reproductive effects in chickens. The associated LOAEL from that study is 12.1 mg/kg-day. This NOAEL and LOAEL were used for this BERA (Table B-16).

3.6.4 Mammals

Copper can be lethal to mammals at high doses (Eisler 1998). Copper is lethal in sheep when eaten for extended periods at more than 80 mg/kg diet (equivalent to 5.1 to 10.7 mg/kg-day), more than 238 mg/kg diet in pigs, and more than 4,000 mg/kg diet in rats (equivalent to more than 133 mg/kg-day). Adverse sublethal effects of copper to sensitive mammals occur at dietary levels ranging from 7.9 mg/kg-day in food to 400 mg/L in drinking water. Chronic toxicity of copper sulfate on the reproduction of mink was evaluated by Aulerich et al. (1982). Data from this study were used by Sample et al. (1996) to support development of a NOAEL of 11.7 mg/kg-day and a LOAEL of 15.1 mg/kg-day for kit mortality.

USEPA (2007a) identified 97 studies with data for mammalian test species, which contained 123 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007a) identified a TRV of 5.60 mg/kg-day, as the highest bounded NOAEL below the lowest bounded LOAEL. This result came from a 4-week feeding study in pigs in which reduced growth and mortality were observed. The associated LOAEL from the same study is 9.34 mg/kg-day. This NOAEL and LOAEL were used for this BERA (Table B-10).

3.7 Lead

Lead has no nutritional or biochemical function (NAS 1980). The mechanism by which lead acts is believed to be indirect interference in normal metal-dependent enzyme functions at specific cellular sites, but toxicity can be affected by many physical and biological variables. In controlled studies, lead adversely affects survival, growth, reproduction, development, and metabolism of most species tested (Eisler 1998). In general, organolead compounds are more toxic than inorganic lead compounds, and young, immature organisms are more susceptible to lead's effects (Eisler 1998).

Birds and mammals exhibit lead toxicity as damage to the nervous system, kidneys, liver, sterility, growth inhibition, developmental retardation, and detrimental effects in blood (Eisler 1988). Irreversible central nervous system damage and decreased intelligence at extremely low doses of lead have been observed in mammals (ATSDR 1997). Inhibition of blood δ-aminolevulinic acid dehydratase, an enzyme critical in heme formation, has been observed as a result of exposure to lead in a variety of fish, invertebrates, and birds (USEPA 2000a).

Lead was selected as a $COPC_E$ for benthic invertebrates, reptiles, birds, and mammals (Table B-1).

3.7.1 Benthic Invertebrates

In aquatic environments, dissolved lead is the most toxic form; organolead compounds are much more toxic to aquatic organisms than are inorganic lead compounds (Eisler 1988; USEPA 2000a). The common forms of dissolved lead are lead sulfate, lead chloride, lead hydroxide, and lead carbonate, but the distribution of salts is highly dependent on the pH of the water. Most lead entering surface waters precipitates in sediment as carbonates or hydroxides. Bioavailability from sediment is controlled by the sediment organic content and acid-volatile sulfide (AVS) concentration (USEPA 2000a).

Lead is accumulated by aquatic organisms equally from water and through food (USEPA 2000a). Although methylated lead is rapidly bioaccumulated from the water by trout, there is no evidence that lead biomagnifies in the aquatic environment.

The AWQC for benthic invertebrates are a CCC of 8.1 and a CMC of 210 µg/L for chronic and acute exposure, respectively (USEPA 2009). For sediment, the ER-L for lead is 46.7 mg/kg dw, and the ER-M is 218 mg/kg dw (NOAA 1999). The ERL of 46.7 mg/kg and the ER-M of 218 mg/kg in sediment were used as the TRVs for lead in benthic invertebrates.

3.7.2 Birds

A review of wildlife toxicity studies by Eisler (1988) reports that among sensitive species of birds, survival was reduced at doses of 75 to 150 mg lead(II)/kg bw or 28 mg alkyl lead/kg bw, reproduction was impaired at dietary levels of 50 mg lead(II)/kg, and signs of poisoning were evident at doses as low as 2.8 mg alkyl lead/kg bw.

USEPA (2005c) identified 54 papers containing avian toxicity data; within these there were 57 NOAEL and/or LOAEL results related to survival, growth, or reproduction. The final NOAEL of 1.63 mg/kg-day developed by USEPA (2005c) is the highest bounded NOAEL below the lowest bounded LOAEL, based on a dietary study of reproductive effects in chickens. The minimum bounded LOAEL is 1.94 mg/kg-day, based on reproductive effects observed in a 5-week feeding study in Japanese quail. Therefore, the NOAEL of 1.63 mg/kg-day and the LOAEL of 1.94 mg/kg-day were selected as the TRVs for birds for this BERA (Table B-16).

3.7.3 Mammals

Among sensitive species of mammals, survival was reduced at acute oral doses of lead as low as 5 mg/kg bw in rats, at chronic oral doses of 0.3 mg/kg bw in dogs, and at dietary levels of 1.7 mg/kg bw in horses. USEPA identified 223 individual NOAEL or LOAEL results relevant to survival, growth, or reproduction in mammals exposed to lead in toxicological studies (USEPA 2005c). USEPA derived the TRV from the highest bounded NOAEL below the lowest bounded LOAEL, which was 4.7 mg/kg-day from a 7-week drinking water study that observed growth effects in rats. The minimum bounded LOAEL is 5.0 mg/kg-day, based on reduced growth observed in a 21-day drinking water study in rats. The NOAEL of 4.7 mg/kg-day and the LOAEL of 5.0 mg/kg-day were selected as the TRVs for this BERA (Table B-10).

3.8 Manganese as a $COPC_E$ for Benthic Invertebrates

Manganese was selected as a $COPC_E$ for benthic invertebrates only (Table B-1). No ER-L/ER-M values or AWQC are available for manganese for the evaluation of risks to benthic invertebrates. An ECOTOX record related to the effect of manganese on survival, growth, or reproduction was a report of no effect on growth at 10 µg/L in a 14-day study of Pacific oysters (Watling 1983, as cited in USEPA 2012). An effect level and the percentage of the tested organisms affected were not reported, and the result was not indicated as statistically significant.

Bengtsson (1978) exposed the harpacticoid copepod *Nitocra spinipes* to individual metal chlorides in brackish water for 96 hours and measured an LC_{50} value of 70,000 µg/L for manganese. This LC_{50} was divided by 10 to estimate an NOAEC of 7,000 µg/L for manganese (Table B-14).

3.9 Mercury

Mercury is a toxic, non-essential element (NAS 1980; USEPA 1999a). Common bacteria convert inorganic forms of mercury to organic forms (Matilainen et al. 1991). Inorganic mercury is less toxic than organomercury compounds, with methylmercury being of greatest concern for potential to cause toxicity in birds and mammals. Methylmercury is highly stable and bioaccumulates and biomagnifies in food chains (USEPA 1999a).

The mechanism of mercury toxicity in animals is interference with metabolism and cell division. Mercury binds strongly with sulfhydryl groups causing inhibition or inactivation of proteins containing thiol ligands and ultimately leading to meiotic disturbances (USEPA 1999a). In all vertebrate receptors, the target organs are the kidney and central nervous system.

At low doses to birds and mammals, mercury adversely affects reproduction, growth, and development, behavior, blood and serum chemistry, motor coordination, vision, hearing, histology, and metabolism. In mammals, methylmercury irreversibly damages the central nervous system and can also be teratogenic and mutagenic. For all organisms tested, early developmental stages were the most sensitive to mercury. Numerous biological and abiotic factors modify the toxicity of mercury compounds, sometimes by an order of magnitude or more, but the mechanisms are not clear (Eisler 1987)

Mercury adversely affects reproduction, growth, behavior, metabolism, blood chemistry, osmoregulation, and oxygen exchange in marine and freshwater organisms. Lethal concentrations of total mercury to sensitive, representative organisms varied from 0.1 to 2.0 µg/L for aquatic fauna. Reproduction was inhibited among sensitive species of aquatic organisms at water concentrations of 0.03 to 0.1 µg/L (Eisler 1987).

Mercury was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

3.9.1 Benthic Invertebrates

The ambient water quality criteria for use in evaluation of risks to benthic invertebrates are 0.94 and 1.8 µg/L for chronic and acute exposures to mercury, respectively (USEPA 2009). For benthic invertebrates in sediment, the ER-L is 0.15 mg/kg dw, and the ER-M is 0.71 mg/kg dw (Long et al. 1995). The ER-L of 0.15 mg/kg and the ER-M of 0.71 mg/kg in sediment were used as the TRVs for mercury in benthic invertebrates.

3.9.2 Fish

Windward (2011a) reviewed six studies of mercury toxicity to fish and selected Matta et al. (2001), who exposed mummichog (*Fundulus heteroclitus)* to methylmercuric chloride in food in a multi-generational test of reproductive effects. Exposure to methylmercury reduced male survival in the parental generation, reduced the ability of offspring to successfully reproduce, and altered sex ratios in offspring. The reduced male survival could have been due to increased aggression observed in treated males but not in treated females; the aggression is consistent with neurotoxic effects of methylmercury but might also be due to aquarium confinement. Based on the findings of Matta et al. (2001), the NOAEL is 0.5 mg/kg in food and the LOAEL is 1.9 mg/kg in food (Table B-15).

3.9.3 Birds

Hill and Schaffner (1976) found a NOAEL of 4 mg/kg diet for reproductive effects in Japanese quail. The LOAEL was 8 mg/kg diet for decreased fertility and hatchability of eggs. Using the data reported in this study, Sample et al. (1996) developed a NOAEL intake of 0.45 mg/kg-day and a LOAEL intake of 0.9 mg/kg-day. Heinz (1979) administered methyl mercury dicyandiamide in the diet to 3 generations of mallard duck. This study reported a chronic LOAEL of 0.5 mg/kg diet for decreased production of eggs and ducklings in the third and second generation, respectively, but 0.5 mg/kg diet did not result in adverse effects in the first generation. This study was not selected to support the TRV for mercury because it does not provide an appropriate model for expected environmental conditions: it is very unlikely that birds using the Site would be exposed to the same concentration over three consecutive generations. Therefore, for the purposes of this BERA, 0.5 mg/kg diet was considered to represent a NOAEL for mercury in ducks. Using the consumption rate reported by Heinz (1979) of 0.156 kg food/kg bw-day from the study, the one-generation NOAEL of 0.078 mg/kg-day was derived for this BERA. The LOAEL of 0.9 mg/kg-day for reproductive effects from the Hill and Schaffner (1976) study was selected as the LOAEL for this BERA (Table B-16).

3.9.4 Mammals

Mercury administered to test animals as inorganic salts tends to be less toxic than is organic methylmercury. Wobeser et al. (1976) administered methylmercury chloride in the diet to mink over a period of 93 days. They found a NOAEL of 1.1 mg/kg diet and a LOAEL of 1.8 mg/kg ww diet for mortality, weight loss, and behavioral abnormalities. Sample et al. (1996) used the data from this study, combined with a subchronic–chronic uncertainty factor of 0.1, to calculate a NOAEL of 0.015 mg/kg-day and a LOAEL of 0.025 mg/kg-day, which were selected as the TRVs for mammals for this BERA (Table B-10). Because of the uncertainty factor applied, the highly toxic form of mercury used, and the relatively high sensitivity of mink relative to other mammals, these TRVs are considered to be very conservative.

3.10 Nickel

Nickel was selected as a $COPC_E$ for fish, reptiles, birds, and mammals (Table B-1).

3.10.1 Fish

The AWQC for nickel (USEPA 2009) were not used as the TRV for nickel in fish because they are based largely on responses in freshwater species. At the time the criteria were derived, there were no paired acute and chronic data for marine fish (or other taxa), and the acute-to-chronic ratio used in the calculations seems to greatly overestimate toxicity to marine species (Hunt et al. 2002). An extensive study with several marine species, including a fish, the topsmelt (*Atherinops affinis*), reports results from both acute and chronic toxicity tests. Fish growth was not affected in the chronic test, and the endpoint was mortality. The lower of two chronic effects levels for mortality in topsmelt was 3,240 µg/L. From these and the acute results, an acute-to-chronic ratio of 6.22 is presented by Hunt et al. (2002), which was in close agreement for all marine species tested in the study.

Because of the new information provided by the Hunt et al. (2002) study, the nickel TRV for fish was derived as follows. Each of the species mean acute values for marine fish provided by USEPA (1986) were divided by an uncertainty factor of 10 to estimate corresponding NOECs, and these were combined with the lowest NOEC for topsmelt from Hunt et al. (2002). From these data, a geometric mean was calculated as 3,595 µg/L. All of the data used in this calculation are presented in Table B-17. This estimated NOAEL for fish, set at two significant figures (3,600 µg/L) was used as the nickel TRV for fish (Table B-15).

3.10.2 Birds

USEPA (2007b) identified 11 studies with data for avian test species. In these studies, there were 17 NOAEL and/or LOAEL results for survival, growth, or reproductive endpoints. USEPA (2007b) derived a TRV of 6.71 mg/kg-day based on the geometric mean of the NOAEL values for growth and reproduction. The minimum bounded LOAEL is 11.5 mg/kg-day from a 42-day feeding study in chickens in which growth effects were observed. The NOAEL of 6.71 mg/kg-day and the LOAEL of 11.5 mg/kg-day were used for this BERA (Table B-4).

3.10.3 Mammals

USEPA identified 61 individual NOAEL or LOAEL results relevant to survival, growth, or reproduction in mammals exposed to nickel in toxicological studies (USEPA 2007b). USEPA derived a TRV of 1.7 mg/kg-day based on the highest bounded NOAEL below the lowest bounded LOAEL. This value came from a study in which juvenile mice dosed orally for 35 days exhibited reproductive effects. The minimum bounded LOAEL is 2.71 mg/kg-day from a 35-day study in which mice exposed orally exhibited reproductive effects. The NOAEL of 1.7 mg/kg-day and the LOAEL of 2.71 mg/kg-day were used for this BERA (Table B-10).

3.11 Thallium as a $COPC_E$ for Benthic Invertebrates

Thallium has applications in rodenticides and insecticides (banned in the U.S. since 1975), treatment of skin infections, manufacture of glass and semiconductors, and infrared detectors. It is considered highly toxic to mammals. Thallium was selected as a $COPC_E$ for benthic invertebrates (Table B-1).

Relatively little information on the toxicity of thallium to aquatic life is available. According to USEPA (1986), acute thallium toxicity to aquatic life in saltwater occurs at concentrations as low as 2,130 µg/L. According to a review by Peter and Viraraghavan (2005), thallium kills insects at 2 mg/L, tadpoles at 0.4 mg/L, and fish at 1 mg/L. There are no AWQC for thallium and no ECOTOX records related to benthic invertebrates. A literature search for marine or estuarine water and sediment toxicity data identified no relevant articles. The acute toxicity value of 2,130 µg/L was divided by 10 to estimate a NOAEC of 213 µg/L in surface water, which was used as the TRV for thallium in benthic invertebrates (Table B-14).

3.12 Vanadium

Vanadium can exist in many valence states (most often 5+) and is common in the earth's crust. It is used in ferrous metallurgy in the manufacture of special steels. Alloys of vanadium with non-ferrous metals are used in aircraft and space technology. Sources to the environment include combustion of fossil fuels and disposal of coal wastes and flyash.

Vanadium was selected as a $COPC_E$ for benthic invertebrates, reptiles, and birds (Table B-1).

3.12.1 Benthic Invertebrates

No ER-L/ER-M values or AWQC are available for vanadium for use in evaluation of risks to benthic invertebrates. There are no ECOTOX records related to survival, growth, or reproduction in benthic invertebrates. The Concise International Chemical Assessment Document for vanadium[5] reports that 50 µg/L is associated with impaired development of oyster (*Crassostrea gigas)* larvae following acute exposures of 48 hours (WHO 2001). This

[5] http://www.inchem.org/documents/cicads/cicads/cicad29.htm#_29ci1A10

was the most sensitive species for which data are available. Development of urchin (*Paracentrotus lividus*) larvae was impaired at 100 µg/L but not at 50 µg/L; and mortality was observed in brine shrimp (*Artemia salina*) after 8-day exposures at 250 µg/L. The acute effect level in oysters of 50 µg/L was divided by 5 to estimate a LOAEC of 10 µg/L, and divided by 10 to estimate a NOAEC of 5 µg/L in marine water. These were used as the TRVs for vanadium in benthic invertebrates (Table B-14).

3.12.2 Birds

Several studies report on the toxicity of vanadium to birds (USEPA 2005d), many of which address survival, growth, and reproductive endpoints. However, the majority of data are for the chicken, with only two studies reporting toxicity to ducks, and one to Japanese quail. USEPA (2005d) identified a TRV of 0.344 mg/kg-day based on the highest bounded NOAEL below the lowest bounded LOAEL for a survival, growth, or reproductive endpoint. This value came from a 5-week feeding study that observed reduced growth in chickens. The lowest bounded LOAEL is 0.413, from a 7-day feeding study in which reproductive effects were observed in chickens. The NOAEL of 0.344 mg/kg-day and the LOAEL of 0.413 mg/kg-day were used for this BERA (Table B-16).

3.13 Zinc

Zinc is required for normal growth, development, and function in all animal species that have been studied (NAS 1980). Zinc attaches to organic molecules such as amino acids, proteins, and nucleic acids, directly binding to sulfhydryl, amino, imidazole, and phosphate groups (NAS 1980). Zinc has low toxicity to birds and mammals, and is not a highly mobile element in aquatic food webs and does not biomagnify in marine or freshwater food webs (USEPA 2000a). Exposures to high concentrations of zinc may result in reduced growth, anemia, reduced bone ash, decreased tissue concentrations of iron, copper, and manganese, and decreased use of calcium and phosphorus (NAS 1980).

Zinc in the water column can partition to dissolved and particulate organic carbon. Water hardness (i.e., calcium concentration), pH, and metal speciation are important factors in controlling the water column concentrations of zinc since the divalent zinc ion is believed to be responsible for observed biological effects (USEPA 2000a). Significant adverse effects of zinc on survival, growth, and reproduction occur in sensitive species of aquatic plants, protozoans, sponges, molluscs, crustaceans, echinoderms, fish, and amphibians at nominal water concentrations between 10 and 25 µg/L (Eisler 1993).

Zinc was selected as a $COPC_E$ for benthic invertebrates, fish, reptiles, birds, and mammals (Table B-1).

3.13.1 Benthic Invertebrates

Bioavailability of zinc in sediments is controlled by the AVS concentration. The ER-L for zinc is 150 mg/kg dw and the ER-M is 410 mg/kg dw (NOAA 1999). The ER-L and ER-M were used as the TRVs for zinc in benthic invertebrates.

3.13.2 Fish

In general, zinc is more toxic to fish embryos and juveniles than to adult fish. Windward (2011b) derived a dietary NOAEL of 1,900 mg/kg based on a 60-day study that observed growth and survival in rainbow trout fry (Mount et al. 1994) and a dietary LOAEL of 2,000 mg/kg based on a 6-week study that observed growth in rainbow trout fingerling (Takeda and Shimma 1977). These values were selected for this BERA (Table B-15).

3.13.3 Birds

Growth of domestic poultry and wild birds was reduced at concentrations in the diet >2,000 mg/kg, and survival was reduced at concentrations >3,000 mg/kg in diet, or at a single oral dose >742 mg/kg bw. Younger stages (i.e., chicks, ducklings) were least resistant (Eisler 1993). A study of dietary exposure of white Leghorn hens to zinc sulfate for 44 weeks found a NOAEL of 228 mg/kg diet and a LOAEL of 2,028 mg/kg diet for decreased egg hatchability. Sample et al. (1996) used data from this study to develop a NOAEL intake of 14.5 mg/kg-day and a LOAEL intake of 131 mg/kg-day.

USEPA (2007c) found 53 studies with data for avian test species. Within these studies, there were 94 NOAEL and/or LOAEL results related to survival, growth, or reproduction, all of which were for domestic poultry. USEPA (2007c) calculated a TRV of 66.1 mg/kg-day based on the geometric mean of NOAEL values for growth and reproduction, but NOAELs ranged as high as 741 mg/kg-day (survival in turkeys), 367 mg/kg-day (reproduction in chickens), and 106 mg/kg-day (growth of chickens). The lowest bounded LOAEL for a growth endpoint was 86.6 mg/kg-day, based on a 14-day feeding study that observed growth reduction in Japanese quail. These values were selected as TRVs for birds (Table B-16).

3.13.4 Mammals

Sensitive species of livestock and small laboratory animals are adversely affected at 90 to 300 mg/kg diet, >90 mg/kg-day repeated oral dose, >300 mg/L drinking water, and >350 mg/kg bw single oral dose. A study of dietary exposure of rats to zinc oxide during gestation reported a NOAEL of 2,000 mg/kg diet and a LOAEL of 4,000 mg/kg diet for increased rates of fetal resorption and reduced fetal growth rates (Schlicker and Cox 1968). Sample et al. (1996) used data from this study to develop a NOAEL intake of 160 mg/kg-day and a LOAEL intake of 320 mg/kg-day.

USEPA's (2007c) literature review found 99 studies with data for mammalian test species. These studies contained 104 NOAEL and/or LOAEL results related to survival, growth, or reproduction. USEPA (2007c) calculated a TRV of 75.4 mg/kg-day, based on the geometric mean of the NOAEL values for growth and reproduction. The lowest bounded LOAEL for a survival, growth, or reproductive endpoint is 75.9 mg/kg-day, based on a 14-week feeding study of reproductive effects in cattle. The NOAEL of 75.4 mg/kg-day and the LOAEL of 75.9 mg/kg bw-day were used in this BERA (Table B-10).

4 REFERENCES

Adams, W.J., G.M. Degraeve, T.D. Sabourin, J.D. Cooney, and G.M. Mosher, 1986. Toxicity and bioconcentration of 2,3,7,8-TCDD to fathead minnow (*Pimephales promelas*). *Chemosphere* 15:1503-1511.

Anchor QEA and Integral, 2010. Final Remedial Investigation/Feasibility Study Work Plan, San Jacinto Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, LLC, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. November 2010.

Andreasen, E.A., L.K. Mathew, C.V. Lohr, R. Hasson, and R.L. Tanguay, 2007. Aryl hydrocarbon receptor activation impairs extracellular matrix remodeling during zebra fish fin regeneration. *Toxicological Sciences* 95(1):215-226.

Ankley, G.T., D.E. Tillitt, J.P. Giesy, P.D. Jones, and D.A. Verbrugge, 1991. Bioassay-derived 2,3,7,8-tetrachlorodibenzo-*p*-dioxin equivalents in PCB-containing extracts from the flesh and eggs of Lake Michigan chinook salmon (*Oncorhynchus tshawytscha*) and possible implications for reproduction. *Can. J. Fish. Aquat. Sci.* 48:1685–1689.

Ashley, C., M.G. Simpson, D.M. Holdich, and D.R. Bell, 1996. 2,3,7,8-tetrachloro-dibenzo-*p*-dioxin is a potent toxin and induces cytochrome P450 in the crayfish, (*Pacifastacus leniusculus). Aquatic Toxicology* 35:157-169.

ATSDR, 1997. Toxicological Profile for Lead. Available at http://www.atsdr.cdc.gov/toxpro2.html. Agency for Toxic Substances and Disease Registry, Division of Toxicology, Toxicology Information Branch, Atlanta, GA.

ATSDR, 2002. Toxicological Profile for DDT, DDE, DDD. U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry, Atlanta, GA.

Aulerich, R.J., and R.K. Ringer, 1977. Current status of PCB toxicity to mink, and effect on their reproduction. *Arch. Environ. Contam. Toxicol.* 6(2-3):279–292.

Aulerich, R.J., R.K. Ringer, M.R. Bleavins, and A. Napolitano, 1982. Effects of supplemental dietary copper on growth, reproductive performance and kit survival of standard dark mink and the acute toxicity of copper to mink. *J. Animal Sci.* 55(2):337–343.

Barber, T.R., D.J. Chappie, D.J. Duda, P.C. Fuchsman, and B.L. Finley, 1998. Using a spiked sediment bioassay to establish a no-effect concentration for dioxin exposure to the amphipod (*Ampelisca abdita*). *Environmental Toxicology and Chemistry* 17:420-424.

Becker, D.S., and T.C. Ginn, 2008. Critical evaluation of the sediment effect concentrations for polychlorinated biphenyls. *Integ. Environ. Assess. Manage.* 4(2):156-170.

Bello, S.M., W. Heideman, and R.E. Peterson, 2004. 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin Inhibits Regression of the Common Cardinal Vein in Developing Zebrafish. *Toxicological Sciences* 78:258-256.

Bengtsson, B.E., 1978. Use of a harpacticoid copepod in toxicity tests. *Mar. Pollut. Bull.* 9:238-241.

Bengtsson, B.E., 1980. Long-term effects of PCB (Clophen A50) on growth, reproduction, and swimming performance in the minnow (*Phoxinus phoxinus*). *Water Research* 14:681-687.

Beresford, W.A., M.P. Donovan, J.M. Henninger, and M.P. Waalkes, 1981. Cadmium, lead in the bones and soft tissues of box turtles caught near smelters. *Bull. Environ. Contam. Toxicol.* 27(1):349–352.

Bergeron, J.M., D. Crews, and J.A. McLachlan, 1994. Polychlorinated biphenyls as environmental estrogens: turtle sex determination as a biomarker of environmental contamination. *Environ. Health Perspect.* 102(9):780–781.

Bishop, C.A., R.J. Brooks, J.H. Carey, P. Ng, R.J. Norstrom, and D.R.S. Lean, 1991a. The case for a cause-effect linkage between environmental contamination and development in eggs of the common snapping turtle (*Chelydra S. serpentina*) from Ontario, Canada. *J. Toxicol. Environ. Health.* 33:521–547.

Bishop, C.A., P. Ng, K.E. Pettit, S.W. Kennedy, J.J. Stegeman, R.J. Norstrom, and R.J. Brooks, 1998. Environmental contamination and developmental abnormalities in eggs and hatchlings of the common snapping turtle (*Chelydra serpentina serpentina*) from the Great Lakes-St Lawrence River basin (1989-91). *Environmental Pollution* I101:143-156.

Bleavins, M.R., R.J. Aulerich, and R.K. Ringer, 1980. Polychlorinated biphenyls (Aroclors 1016 and 1242): effects on survival and reproduction in mink and ferrets. *Arch. Environ. Contam. Toxicol.* 9(5):627–635.

Bowman, R.E., S.L. Schantz, M.L. Gross, and S.A. Ferguson, 1989. Behavioral effects in monkeys exposed to 2,3,7,8-TCDD transmitted maternally during gestation and for four months of nursing. *Chemosphere* 18(1-6):235–242.

Broyles, R.H., and M.I. Noveck, 1979. Uptake and distribution of 2,4,5,2',4',5'-hexachlorobiphenyl in fry of lake trout and Chinook salmon and its effects on viability. *Toxicology and Applied Pharmacology* 50(2):299-308.

Bruggeman, V., O. Onagbesan, L. Dumez, B. De Ketelaere, and E. Decuypere, 2005. Effects of early prenatal exposure to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) on postnatal reproduction in the laying hen (*Gallus gallus*). *Comp Biochem Physiol C Toxicol Pharmacol*. 141(4):349-55.

Butler, R.A., M.L. Kelley, W.H. Powell, M.E. Hahn, and R.J. Van Beneden, 2001. An aryl hydrocarbon receptor homologue from the soft-shell clam, (*Mya arenaria*): evidence that invertebrate AHR homologues lack TCDD and BNF binding. *Gene* 278:223-234.

Butler, R.A., M.L. Kelley, K.E. Olberding, G.R. Gardner, and R.J. Van Beneden, 2004. Aryl hydrocarbon receptor (AhR)-independent effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) on softshell clam (*Mya arenaria*) reproductive tissue. *Comparative Biochemistry and Physiology*, Part C, Toxicology & Pharmacology 138(3):375-381.

Carney, S.A., A.L. Prasch, W. Heideman, and R.E. Peterson, 2006. Understanding dioxin developmental toxicity using the zebrafish model. *Birth Defects Research* 76(1):7-18.

Cecil, H.C., J. Bitman, R.J. Lillie. G.F. Fries, and J. Verrett, 1974. Embryotoxic and teratogenic effects in unhatched fertile eggs from hens fed polychlorinated biphenyls (PCBs). *Bull. Environ. Contam. Toxicol.* 11(6): 489-455.

Chapman, P.M., A. Fairbrother, and D. Brown, 1999. A critical evaluation of safety (uncertainty) factors for ecological risk assessment. *Environ. Toxicol. Chem.* 17(1):99-108.

Cohen-Barnhouse, A.M., M.J. Zwiernik, J.E. Link, S.D. Fitzgerald, S.W. Kennedy, J.P. Giesy, S. Wiseman, P.D. Jones, J.L. Newsted, D. Kay, and S.J. Bursian, 2011. Developmental and posthatch effects of *in ovo* exposure to 2,3,7,8-TCDD, 2,3,4,7,8-PeCDF, and 2,3,7,8-TCDF in Japanese quail (*Coturnix japonica*), common pheasant (*Phasianus colchicus*), and white leghorn chicken (*Gallus gallus domesticus*) embryos. *Environmental Toxicology and Chemistry* 30(7):1649-1668.

Cooper, K.R., and M.L. Wintermyer, 2009. A critical review: 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) effects on gonad development in bivalve mollusks. *Journal of Environmental Science and Health,* Part C, Environmental, Carcinogenesis & Ecotoxicology Reviews 27(4):226-245.

Custer, T.W., and G.H. Heinz, 1980. Reproductive success and nest attentiveness of mallard ducks fed Aroclor. *Environ. Pollut.* 21:313–318.

Custer, C.M., T.W. Custer, P.D. Allen, K.L. Stromborg, and M.J. Melancon, 1998. Reproduction and environmental contamination in tree swallows nesting in the Fox River drainage and Green Bay, Wisconsin, USA. *Environ. Toxicol. Chem.* 17(9):1786–1798.

Custer, C.M., T.W. Custer, C.J. Rosiu, M.J. Melancon, J.W. Bickham, and C.W. Matson, 2005. Exposure and effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in tree swallows (*Tachycineta bicolor*) nesting along the Woonasquatucket River, Rhode Island, USA. *Environmental Toxicology and Chemistry* 24(1):93-109.

Custer, T.W., C.M. Custer, R.K. Hines, and D.W. Sparks, 2010. Trace elements, organochlorines, polycyclic aromatic hydrocarbons, dioxins, and furans in lesser scaup wintering on the Indiana Harbor Canal. *Environmental Pollution* 110:469-482.

Cuthill, I.C., S.A. Maddocks, C.V. Weall, and E.K.M. Jones, 1999. Body mass regulation in response to changes in feeding predictability and overnight energy expenditure. *Behav. Ecol.* 11(2):189-195.

Dahlgren, R., R. Linder, and C. Carlson, 1972. Polychlorinated biphenyls: their effects on penned pheasants. *Environ. Health Perspect.*1:89-101. (As cited by Sample et al, 1996)

David, R.M., M.R. Moore, D.C. Finney et al., 2000. Chronic toxicity of di(2-ethylhexyl)phthalate in rats. *Toxicol. Sci.* 55:433-443 (as cited in ATSDR 2002b).

De Solla, S.R., C.A. Bishop, K.G. Van der, and R.J. Brooks, 1998. Impact of organochlorine contamination on levels of sex hormones and external morphology of common snapping turtles (*Chelydra serpentina serpentina*) in Ontario, Canada. *Environ. Health Perspect.* 106(5):253–260.

Eisler, R., 1985. Cadmium Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85 (1.8). U.S. Department of the Interior, U.S. Fish and Wildlife Service, Laurel, MD.

Eisler, R., 1986a. Dioxin Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. May 1986. Biological Report 85 (1.8). Patuxent Wildlife Research Center, U.S. Fish and Wildlife Service, Laurel, MD 20708.

Eisler, R., 1986b. Polychlorinated Biphenyl Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85 (1.7). U.S. Department of the Interior, U.S. Fish and Wildlife Service, Laurel, MD.

Eisler, R., 1987. Mercury Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85 (1.14). U.S. Department of the Interior, U.S. Fish and Wildlife Service, Laurel, MD.

Eisler, R., 1988. Lead Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 85 (1.10). U.S. Department of the Interior, U.S. Fish and Wildlife Service, Laurel, MD.

Eisler, R., 1993. Zinc Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. Biological Report 10. U.S. Department of the Interior, U.S. Fish and Wildlife Service, Laurel, MD.

Eisler, R., 1998. Copper Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. USGS/BRD/BSR-1998-0002. U.S. Geological Survey, Biological Resources Division, Reston, VA.

Eisler, R., and A.A. Belisle, 1996. Planar PCB Hazards to Fish, Wildlife, and Invertebrates: A Synoptic Review. National Biological Service: U.S Department of the Interior, Washington, DC.

Elliott, J.E., R.J. Norstrom, and G.E. Smith, 1996. Patterns, trends, and toxicological significance of chlorinated hydrocarbon and mercury contaminants in bald eagle eggs from the Pacific coast of Canada, 1990-1994. *Arch. Environ. Contam Toxicol* 31(3):354–367.

Elliott, J.E., M.L. Harris, L.K. Wilson, P.E. Whitehead, and R.T. Norstrom, 2001. Monitoring temporal and spatial trends in polychlorinated dibenzo-*p*-dioxins (PCDDs) and dibenzofurans (PCDFs) in eggs of great blue heron (*Ardea herodias*) on the Coast of British Columbia, Canada, 1983-1998. *Ambio.* 30(7):416-428.

Elonen, G.E., R.L. Spehar, G.W. Holcombe, R.D. Johnson, J.D. Fernandez, R.J. Erickson, et al., 1998. Comparative toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to seven freshwater fish species during the early life-stage development. *Environ. Toxicol. Chem.* 17(3):472–483.

Faqi, A.S., P.R. Dalsenter, H.-J. Merker, and I. Chahoud, 1998. Reproductive toxicity and tissue concentrations of low doses of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in male offspring rats exposed throughout pregnancy and lactation. *Toxcicol. Appl. Pharmacol.* 150:138-392.

Fernie, K., J. Smiths, and G. Bortolotti, 2003a. Developmental toxicity of *in ovo* exposure to polychlorinated biphenyls: I. Immediate and subsequent effects on first-generation nestling American kestrels (*Falco sparverius*). *Environ. Toxicol. Chem.* 22(3):554-560.

Fernie, K., G. Bortolotti, K. Driuillard, J. Smiths, and T. Marchant, 2003b. Developmental toxicity of in ovo exposure to polychlorinated biphenyls: II. Effects of maternal or paternal exposure on second-generation nestling American kestrels (*Falco sparverius*). *Environ. Toxicol. Chem.* 22(11):2688-2694.

Fernie, K., G. Bortolotti, and J. Smiths, 2003c. Reproductive abnormalities, teratogenicity, and developmental problems in American kestrels (*Falco sparverius*) exposed to polychlorinated biphenyls. *J. Toxicol. Environ. Health.* 66:2089-2103.

Frank, D.S., M.A. Mora, J.L. Sericano, A.L. Blankenship, K. Kannan, and J.P. Giesy, 2001. Persistent organochlorine pollutants in eggs of colonial waterbirds from Galveston Bay and East Texas, USA. *Environmental Toxicology and Chemistry* 20(3):608-617.

Fredricks, T.B., J.P. Giesy, S.J. Coefield, R.M. Seston, D.L. Tazelaar, S.A. Roark, D.P. Kay, J.L. Newsted, and M.J. Zwiernik, 2011. Multiple lines of evidence risk assessment of terrestrial passerines exposed to PCDFs and PCDDs in the Tittabawassee River Floodplain, Midland, Michigan, USA. *Human and Ecological Risk Assessment* 17:159-186.

Gale, R.W., J.M. Bergeron, E.J. Willingham, and D. Crews, 2002. Turtle sex determination essay: Mass balance and responses to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and 3,3',4,4',5-pentachlorobiphenyl. *Environ Toxicol Chem.* 21(11):2477-82.

Giesy, J.P., and K. Kannan, 1998. Dioxin-like and non-dioxin-like toxic effects of polychlorinated biphenyls (PCBs): implications for risk assessment. *Crit. Rev. Toxicol.* 28:511-569.

Gilbertson, M., T. Kubiak, J. Ludwig, and G. Fox, 1991. Great Lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. *J. Toxicol. Environ. Health* 33(4):455–520.

Golub, M.S., J.M. Donald, and J.A. Reyes, 1991. Reproductive toxicity of commercial PCB mixtures: LOAELs and NOAELs from animal studies. *Environ. Health. Perspect.* 94:245-253.

Hahn, M.E., A. Poland, E. Glover, and J.J. Stegeman, 1992. The Ah receptor in marine animals: Phylogenetic distribution and relationship to cytochrome P4501A inducibility. *Marine Environmental Research* 34:87-92.

Hamm, J.T., C.-Y. Chen, and L.S. Birnbaum, 2003. A mixture of dioxins, furans, and non-ortho PCBs based upon consensus toxic equivalency factors produces dioxin-like reproductive effects. *Toxicological Sciences* 74:182-191.

Hansen, D.J., S.C. Schimmel, and J. Forester, 1973. Aroclor 1254 in eggs of sheepshead minnows: Effects on fertilization success and survival of embryos and fry. Proceedings 27th Annual Conference of the Southeastern Association of Game Fish Commission, p. 420-426.

Hatakayama, S., and M. Yasuno, 1987. Chronic effects of Cd on the reproduction of the guppy (*Poecilia reticulata*) through Cd-accumulated midge larvae (*Chironomus yoshimatsui*). *Ecotox Environ Saf* 14:191-207 (as cited in Windward 2011b).

Head, J.A., M.E. Hahn, and S.W. Kennedy, 2008. Key amino acids in the aryl hydrocarbon receptor predict dioxin sensitivity in avian species. *Environ. Sci. Technol.* 42:7535-7541.

Heath, R.G., J.W. Spann, and J.F. Kreitzer, 1969. Marked DDE impairment of mallard reproduction in controlled studies. *Nature* 224:47–48.

Heath, R.G., J.W. Spann, J.F. Kreitzer, and C. Vance, 1972. Effects of polychlorinated biphenyls on birds. In: Symposium on Chemical Pollution. XV Cong. of Int. Ornithol. Den Haag. 475-485.

Heaton, S.N., S.J. Bursian, J.P. Giesy, D.E. Tillitt, J.A. Render, P.D. Jones, et al., 1995a. Dietary exposure of mink to carp from Saginaw Bay, Michigan: 2. Hematology and liver pathology. *Arch. Environ. Contam. Toxicol.* 29(3):411–417.

Heaton, S.N., S.J. Bursian, D.E. Tillitt, J.A. Render, P.D. Jones, D.A. Verbrugge, et al., 1995b. Dietary exposure of mink to carp from Saginaw Bay, Michigan. 1. Effects on reproduction and survival, and the potential risks to wild mink population. *Arch. Environ. Contam. Toxicol.* 28:334–343.

Hecker, M., M.B. Murphy, J.P. Giesy, and W.A. Hopkins, 2006. Induction of cytochrome P4501A in African brown house snake (*Lamprophis fuliginosus*) primary hepatocytes. *Environmental Toxicology* 25(2):496-502.

Heiden, T.K., M.J. Carvan III, and R.J. Hutz, 2006. Inhibition of follicular development, vitellogenesis, and serum17b-estradiol concentrations in zebrafish following chronic, sublethal dietary exposure to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin. *Toxicological Sciences* 90(2):490-499.

Heinz, G.H., 1979. Methyl mercury: reproductive and behavioral effects on three generations of mallard ducks. *J. Wildl. Manage.* 43:394–401.

Heitmuller, P.T., T.A. Hollister, and P.R. Parrish, 1981. Acute toxicity of 54 industrial chemicals to sheepshead minnows (*Cyprinodon variegatus*). *Bull. Environ. Contam. Toxicol.* 27(5): 596-604 (as cited in USEPA 2012).

Henshel, D.S., B. Hehn, R Wagey, M. Vo, and J.D. Steevens, 1997a. The Relative Sensitivity of Chicken Embryos to Yolk- or Air-Cell-Injected 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin. *Environmental Toxicology and Chemistry* 16(4):725-732.

Henshel, D.S., J.W. Martin and J.C. Dewitt. 1997b. Brain asymmetry as a potential biomarker for developmental TCDD intoxication: A dose-response study. *Environ. Health Perspect.* 105: 718-725.

Hill, E.F., and C.S. Shaffner, 1976. Sexual maturation and productivity of Japanese quail fed graded concentrations of mercuric chloride. *Poult. Sci.* 55(4):1449–1459.

Ho, K.T., R.A. McKinney, A. Kuhn, M.C. Pelletier, and R.M. Burgess, 1997. Identification of acute toxicants in New Bedford Harbor sediments. *Environmental Toxicology and Chemistry* 16(3):551-558.

Hochstein, J.R., S.J. Bursian, and R.J. Aurelich, 1998. Effects of dietary exposure to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in adult female mink (*Mustela vison*). *Arch. Env. Contam.* 35(2):348-353.

Hunt, J.W., B.S. Anderson, B.M. Phillips, R.S. Tjeerdema, H.M. Puckett, M. Stephenson, D.W. Tucker, and D. Watson, 2002. Acute and chronic toxicity of nickel to marine organisms: Implications for water quality criteria. *Environmental Toxicology and Chemistry.* 21(11):2423-2430.

Isensee, A.R., and G.E. Jones, 1975. Distribution of 2,3,7,8-tetrachlorodibenzo-p-dioxin (TCDD) in aquatic model ecosystem. *Environ. Sci. Technol.* 9(7):668–672.

Ikeda, M., M. Tamura, J. Yamashita, C. Suzuki, and T. Tomita, 2005. Repeated *in utero* and lactational 2,3,7,8-tetrachlorodibenzo-*p*-dioxin exposure affects male gonads in offspring, leading to sex ratio changes in F2 progeny. *Toxicol Appl Pharmacol* 206(3):351-355.

Integral, 2012. Draft Exposure Assessment Memorandum, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. January.

Johnson, W.W., and M.T. Finley, 1980. Handbook of acute toxicity of chemicals to fish and aquatic invertebrates. Summaries of toxicity tests conducted at Columbia National Fisheries Research Laboratory, 1965-78. U.S. Fish Wildl. Serv. Resour. Publ. 137. 98p.

Johnson, R.D., J.E. Tiege, K.M. Jensen, J.D. Fernandez, A.L. Linnum, D.B. Lothenbach, et al., 1998. Toxicity of 2,3,7,8-tetrachlorodibenzo-p-dioxin to early life stage brook trout (*Salvelinus fontinalis*) following parental dietary exposure. *Environ. Toxicol. Chem.* 17:2408–2421.

Karchner, S.I., D.G. Franks, S.W. Kennedy and M.E. Hahn, 2006. The Molecular Basis for Differential Dioxin Sensitivity in Birds: Role of the Aryl Hydrocarbon Receptor.

Proceedings of the National Academy of Sciences of the United States of America. 103(16):6252-6257.

Keller, J.M., J.R. Kucklick, M.A. Stamper, C.A. Harms, and P.D. Clellan-Green, 2004. Associations between organochlorine contaminant concentrations and clinical health parameters in loggerhead sea turtles from North Carolina, USA. *Environ. Health Perspect.* 112(10):1074–1079.

Khera, K.S., and J.A. Ruddick, 1973. Polychlorodibenzo-p-dioxins: Perinatal effects and the dominant lethal test in Wistar rats. In: *Chlorodioxins - origin and fate: Advances in Chemistry Series 120*. E.H. Blair (ed). American Chemical Society, Washington, DC.

Kim, J.S., and P. Chin, 1995. Acute and chronic toxicity of phenol to the mysid, *Archaeomysis kokuboi. Journal of Korean Fisheries Society* 28(1):87-97

Kociba, R.J., D.G. Keyes, J.E. Beyer, R.M. Carreon, C.E. Wade, D.A. Dittenber, et al., 1978. Results of a two-year chronic toxicity and oncogenicity study of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in rats. *Toxicol Appl. Pharmacol.* 46(2):279–303.

Kosutzky, J., O. Adamec, and E. Bobáková, 1979. Effects of polychlorinated biphenyls on poultry reproduction. *Bull. Environ. Contam. Toxicol.* 21:737-742.

Kubiak, T.J., and D.A. Best, 1991. Wildlife Risks Associated with Passage of Contaminated, Anadromous Fish at Federal Energy Regulatory Commission Licensed Dams in Michigan. Division of Ecological Services, Contaminants Program, East Lansing, MI.

Kubiak, T.J., H.J. Harris, L.M. Smith, T.R. Schwarts, D.L. Stalling, J.A. Trick, et al., 1989. Microcontaminants and reproductive impairment of Forster's Tern on Green Bay, Lake Michigan-1983. *Arch. Environ. Contam. Toxicol* 18:706–727.

Lillie, R.J., H.C. Cecil, J. Bitman, G.G. Fries, and J. Verrett, 1974. Differences in response of caged white leghorn layers to various polychlorinated biphenyls (PCBs) in the diet. *Poul. Sci.* 53:726-732.

Linder, G., and B. Grillitsch, 2000. *Ecotoxicology of Amphibians and Reptiles.* Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Linzey, A.V., 1987. Effects of chronic polychlorinated biphenyls exposure on reproductive success of white-footed mice (*Peromyscus leucopus*). *Arch. Environ. Contam. Toxicol.* 16(4):455-460.

Linzey, A.V., 1988. Effects of chronic polychlorinated biphenyls exposure on growth and reproduction of second generation white-footed mice (*Peromyscus leucopus*). Arch*. Environ. Contam. Toxicol.* 17: 39-45.

Long, E.R., D.D. MacDonald, S.L. Smith, and F.D. Calder, 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. *Environmental Management* 19(1):81-97.

Loonen, H., C. van de Guchte, J.R. Parsons, P. de Voogt, and H.A.J. Govers, 1996. Ecological hazard assessment of dioxins: Hazards to organisms at different levels of aquatic food webs (fish-eating birds and mammals, fish and invertebrates). *Science of the Total Environment* 182:93-103.

Mac, M.J., 1988. Toxic substances and survival of Lake Michigan salmonids: Field and laboratory approaches. In: *Toxic Contaminants and Ecosystem Health: A Great Lakes Focus*. M.Evans (ed). John Wiley and Sons, New York, NY. pp. 389–401.

Matilainen, T., M. Vetra, M. Niemi, and A. Uusi-Rauva, 1991. Specific rates of net methylmercury production in lake sediments. *Water Air Soil Pollut.* 56:595–605.

Matta, M.B., J. Linse, C. Cairncross, L. Francendese, and R.M. Kocan, 2001. Reproductive and transgenerational effects of methylmercury of Aroclor 1268 on *Fundulus heteroclitus*. *Environ. Toxicol. Chem.* 20(2):327-335.

Mayer, F.L., P.M. Mehrle, and H.O. Sanders, 1977. Residue dynamics and biological effects of polychlorinated biphenyls in aquatic organisms. *Arch. Environ. Contam Toxicol* 5(4):501–511.

McCoy, G., M.F. Finlay, A. Rhone, K. James, and G.P. Cobb, 1995. Chronic polychlorinated biphenyls exposure on three generations of oldfield mice (*Permyscus polionotus*): effects on reproduction, growth, and body residues. *Arch. Environ. Contam. Toxicol.* 28:431-435.

McFarland, V.A., and J.U. Clarke, 1989. Environmental occurrence, abundance, and potential toxicity of polychlorinated biphenyl congeners: Considerations for a congener specific analysis. *Environ. Health Perspect.* 81:225–239.

Mehring, A.L. Jr., J.H. Brumbaugh, and A.J. Sutherland, 1960. The tolerance of growing chickens for dietary copper. *Poult. Sci.* 39:713–719.

Mehta, V., R.E. Peterson, and W. Heideman, 2008. 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin exposure prevents cardiac valve formation in developing zebrafish. *Toxicological Sciences 104(2):303-311.*

Miller, R.A., L.A. Norris, and C.L. Hawkes, 1973. Toxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) in aquatic organisms. *Environmental Health Perspectives* 5:177-186.

Moore, J.N., J.L. Newsted, M. Hecker, M.J. Zwiernik, S.D. Fitzgerald, D.P. Kay, X. Zhang, E.B. Higley, L.L. Aylward, K.J. Beckett, R.A. Budinsky, S.J. Bursian, and J.P. Giesy, 2009. Hepatic P450 enzyme activity, tissue morphology and histology of mink (*Mustela vison*) exposed to polychlorinated dibenzofurans. *Arch Environ Contam Toxicol.* 57:416-425.

Mount, D.R., A.K. Barth, T.D. Garrison, K.A. Barten, and J.R. Hockett, 1994. Dietary and waterborne exposure of rainbow trout (*Oncorhynchus mykiss*) to copper, cadmium, lead and zinc using a live diet. *Environ. Toxicol. Chem.* 13(12):2031-41.

Murray, F.J., F.A. Smith, K.D. Nitschke, C.G. Humiston, R.J. Kociba, and B.A. Schwetz, 1979. Three generation reproduction study of rats given 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) in the diet. *Toxicology and Applied Pharmacology* 50:241–252.

Mutoh, J., J. Taketoh, K. Okamura, T. Kagawa, T. Ishida, Y. Ishii, and H. Yamada, 2006. Fetal pituitary gonadotropin as an initial target of dioxin in its impairment of cholesterol transportation and steroidogenesis in rats. *Endocrinology* 147(2):927-936.

Nacci, D.E., D. Champlin, L. Coiro, R. McKinney, and S. Jayaraman, 2002. Predicting the occurrence of genetic adaptation to dioxinlike compounds in populations of the estuarine fish *Fundulus heteroclitus*. *Environmental Toxicology and Chemistry* 21(7):1525-1532.

NAS, 1980. Mineral tolerance of domestic animals. National Academy of Sciences, National Academy Press, Washington, DC.

Nebeker, A. V., and F. A. Puglisi, 1974. Effect of polychlorinated biphenyls (PCBs) on survival and reproduction of *Daphnia, Gammarus, and Tanytarsus. Trans. Amer. Fish. Soc.* 103: 722-728.

Nichols, M., J. Pruitt, D. Munsey, G. Good, B. Meyer, K. Urban, K. Francl, and P. DiLaura, 1999. *Macrochelys temminckii* (On-line), Animal Diversity Web. Accessed

October 11, 2011 http://animaldiversity.ummz.umich.edu/site/accounts/information/Macrochelys_temminckii.html.

Niimi, A.J., 1996. PCBs in Aquatic Organisms. In: *Environmental Contaminants in Wildlife*. W.N. Beyer, G.H. Heinz, A.W. Redmon-Norwood (eds). Lewis Publishers, Boca Raton. pp. 117–151.

NOAA, 1999. National Status and Trends Program. Sediment Quality Guidelines. Available at http://response.restoration.noaa.gov/book_shelf/121_sedi_qual_guide.pdf. National Oceanic and Atmospheric Administration, Seattle, WA.

Nosek, J.A., S.R. Craven, J.R. Sullivan, J.R. Olsen, and R.E. Peterson, 1992a. Metabolism and disposition of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in ring-necked pheasant hens, chicks, and eggs. *Journal of Toxicology and Environmental Health* 35:153-164.

Nosek, J.A., S.R. Craven, J.R. Sullivan, S.S. Hurley, and R.E. Peterson, 1992b. Toxicity and reproductive effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in ring-necked pheasant hens. *Journal of Toxicology and Environmental Health* 35:187-198.

Nosek, J.A., J.R. Sullivan, S.R. Craven, A.G. Gendron-Fitzpatrick, and R.E. Peterson, 1993. Embryotoxicity of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin in the ring-necked pheasant. *Environmental Toxicology and Chemistry* 12:1215-1222.

NYSDEC, 2002. New York State Department of Environmental Conservation (NYSDEC). Onondaga Lake Baseline Ecological Risk Assessment. Prepared by TAMS Consultants, Inc. and YEC, Inc.

O'Shea, T.J., and C.J. Stafford, 1980. Phthalate plasticizers: accumulation and effects on weight and food consumption in captive starlings. *Bull. Environ. Contam. Toxicol.* 25:345-352.

Ohsaka, S., Y. Miyabara, M. Sakaue, R. Ishimura, M. Kakeyama, H. Izumi, J. Yonemoto, and C. Tohyama, 2002. Developmental stage-specific effects of perinatal 2,3,7,8-tetrachlorodibenzo-*p*-dioxin exposure on reproductive organs of male rat offspring. *Toxicological Sciences* 66:283-292.

Orn, S., P.L. Andersson, L. Forlin, M. Tysklind, and L. Norrgren, 1998. The impact on reproduction of an orally administered mixture of selected PCBs in zebrafish (*Danio rerio*). *Arch. Environ. Contam. Toxicol.* 35:52–57.

Pauli, B.D., J.A. Perrault and S.L. Money. 2000. RATL: A Database of Reptile and Amphibian Toxicology Literature. Technical Report Series No. 357. Canadian Wildlife Service, Headquarters, Hull, Québec, Canada. Available at: http://dsp-psd.pwgsc.gc.ca/Collection/CW69-5-357E.pdf.

Peakall, D.B., 1971. Effect of polychlorinated biphenyls (pcb's) on the eggshells of ring doves. *Bull. Environ. Contam. Toxicol.* 6(2): 100-101.

Peakall, D.B., and M.L. Peakall, 1973. Effects of a polychlorinated biphenyl on the reproduction of artificially and naturally incubated dove eggs. *J. Appl. Ecol.* 10(3): 863-868.

Peter, A.L., and T. Viraraghavan, 2005. Thallium: a review of public health and environmental concerns. *Environ. Int.* 31(4):493–501.

Pindur, U., and T. Lemster, 2001. Advances in marine natural products of the indole and annelated indole series: chemical and biological aspects. *Curr. Med. Chem.* 8(13):1681.

Platonow, N.S., and B.S. Reinhart, 1973. The effects of polychlorinated biphenyls (Aroclor 1254) on chicken egg production, fertility and hatchability. *Can. J. Comp. Med.* 37(4):341–346.

Portelli, M.J., and C.A. Bishop, 2000. *Ecotoxicolocgy of Amphibians and Reptiles.* Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.P. Giesy, K.L. Stromborg, and S.J. Bursian, 1996. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126) and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD) injected into the yolks of chicken (*Gallus domesticus*) eggs prior to incubation. *Arch Environ Contam Toxicol.* 31:404-409.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, J.F. Powell, J.C. Restum, K.L. Stromborg, J.P. Giesy, and S.J. Bursian, 1997a. Effects of 3,3',4,4',5-pentachlorobiphenyl (PCB 126), 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), or an extract derived from field-collected cormorant eggs injected into double-crested cormorant (*Phalacrocorax auritus*) eggs. *Environmental Toxicology and Chemistry* 16(7):1450-1455.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, K.L. Stromborg, T.J. Kubiak, J.P. Giesy and S.J. Bursian, 1997b. Organochlorine contaminants in double-crested cormorants from Green Bay, Wisconsin: II. Effects of an Extract Derived from Cormorant Eggs on the Chicken Embryo. *Archives of Environmental Contamination and Toxicology* 32(3):316-322.

Powell, D.C., R.J. Aulerich, J.C. Meadows, D.E. Tillitt, M.E. Kelly, K.L. Stromborg, M.J. Melancon, S.D. Fitzgerald, and S.J. Bursian, 1998. Effects of 3,3',4,4',5-pentachlorobiphenyl and 2,3,7,8-tetrachlorodibenzo-*p*-dioxin injected into the yolks of double-crested cormorant (*Phalacrocorax auritus)* eggs prior to incubation. *Environmental Toxicology and Chemistry.* 17(10):2035-2040.

Pruell, R.J., N.I. Rubinstein, B.K. Taplin, J.A. LiVolsi, and R.D. Bowen, 1993. Accumulation of polychlorinated organic contaminants from sediment by three benthic marine species. *Archives of Environmental Contamination and Toxicology* 24:290-297.

Restum, J.C., S.J. Bursian, J.P. Giesy, J.A. Render, W.G. Helferich, E.B. Shipp, et al., 1998. Multigenerational study of the effects of consumption of PCB-contaminated carp from Saginaw Bay, Lake Huron, on mink. 1. Effects on mink reproduction, kit growth and survival, and selected biological parameters. *J. Toxicol. Environ. Health* 54 Part A:343–375.

Risebrough, R.W., and D.W. Anderson, 1975. Some effects of DDE and PCB on mallards and their eggs. *Journal of Wildlife Management* 39(3):508–513.

Rhodes, L.D., G.R. Gardner, and R.J. van Beneden, 1997. Short-term tissue distribution, depuration and possible gene expression effects of [^{3}H]TCDD exposure in soft-shell clams (*Mya arenaria*). *Environ. Toxicol. Chem.* 16(9):1888-1894.

Roberts, J.R., D.W. Rodgers, J.R. Bailey, and M.A. Rorke, 1978. Polychlorinated biphenyls: biological criteria for an assessment of their effects on environmental quality. National Research Council of Canada. Report No. 16077, 172 pp. (as cited by USEPA, 2000b)

Robinson, K.M., and M.R. Wells. 1975. Retention of a single oral dose of cadmium in tissues of the softshell turtle, *Trionyx spinifer*. *Bull. Environ. Contam Toxicol* 14(6):750–752.

Rubenstein, N.I., R.J. Pruell, B.K. Taplin, J.A. LiVolsi, and C.B. Norwood, 1990. Bioavailability of 2,3,7,8-TCDD, 2,3,7,8-TCDF and PCBs to marine benthos from Passaic River sediments. *Chemosphere* 20:1097-1102.

Safe, S., 1991. Polychlorinated dibenzo-*p*-dioxins and related compounds: Sources, environmental distribution and risk assessment. *Environ. Carcinog. Ecotoxicol. Rev.* C9(2):261–302.

Safe, S.H., and D.R. Phil, 1990. Polychlorinated biphenyls (PCBs), dibenzo-*p*-dioxins (PCDDs), dibenzofurans (PCDFs), and related compounds: environmental and mechanistic considerations which support the development of toxic equivalency factors (TEFs). *Toxicology* 21(1):51–87.

Sager, D.B., and D.M. Girard, 1994. Long-term effects on reproductive parameters in female rats after translactational exposure to PCBs. *Environ. Res.* 66(1):52–76.

Sample, B.E., D.M. Opresko, and G.W.I. Suter, 1996. Toxicological Benchmarks for Wildlife: 1996 Revision. ES/ER/TM-86/R3 Contract DE-AC05-84OR21400. Lockheed Martin Energy Systems, Inc., Oak Ridge National Laboratory, Risk Assessment Program, Health Sciences Research Division, Oak Ridge, TN. Appendix A.

Sanderson, J.T., S.W. Kennedy, and J.P. Giesy, 1998. *In vitro* induction of ethoxyresorufin-*O*-deethylase and porphyrins by halogenated aromatic hydrocarbons in avian primary hepatocytes. *Environ. Toxicol. Chem.* 17:2006-2018

Schantz, S.L., S.A. Ferguson, and R.E. Bowman, 1992. Effects of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin on Behavior of Monkeys in Peer Groups. *Neurotoxicology and Teratology* 14:433-446.

Schlicker, S.A., and D.H. Cox, 1968. Maternal dietary zinc, and development and zinc, iron, and copper content of the rat fetus. *J. Nutr.* 95:287–294.

Schwetz, B.A., J.M. Norris, G.L. Sparschu, U.K. Rowe, P.J. Gehring, J.L. Emerson, et al., 1973. Toxicology of chlorinated dibenzo-*p*-dioxins. *Environ. Health Perspect.* 5:87–99.

Shain, W., B. Bush, and R. Seegal, 1991. Neurotoxicity of polychlorinated biphenyls: structure-activity relationship of individual congeners. *Toxicol Appl. Pharmacol.* 111(1):33–42.

Shelby, J.A., and M.T. Mendonça, 2001. Comparison of reproductive parameters in male yellow-blotched map turtles (*Graptemys flavimaculata*) from a historically contaminated site and a reference site. Department of Biological Sciences, Auburn University, 331 Funchess Hall, 36849, Auburn, AL, USA.

Sparling, D.W., C.A. Bishop, and G. Linder (ed), 2000a. *Ecotoxicology of Amphibians and Reptiles.* Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Sparling, D.W., C.A. Bishop, and G. Linder (ed), 2000b. *Ecotoxicology of Amphibians and Reptiles*. Pensacola, FL: Society of Environmental Toxicology and Chemistry (SETAC). 904 pp.

Spitsbergen, J.M., M.K. Walker, J.R. Olson, and R.E. Peterson, 1991. Pathologic alterations in early life stages of lake trout, *Salvelinus namaycush*, exposed to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin as fertilized eggs. *Aquatic Toxicology* 19:41-72.

Steevens, J.A., M.R. Reiss, and A.V. Pawlisz, 2005. A methodology for deriving tissue residue benchmarks for aquatic biota: a case study for fish exposed to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin and equivalents. *Integrated Environmental Assessment and Management* 1(2):142–151.

Summer C.L., J.P. Giesy, S.J. Bursian, J.A. Render, T.J. Kubiak, P.D. Jones, D.A. Verbrugge, and R.J. Aulerich, 1996. Effects induced by feeding organochlorine-contaminated carp from Saginaw Bay, Lake Huron, to laying White Leghorn hens. I. Effects on health of adult hens, egg production, and fertility. *J. Toxicol. Environ. Health.* 49(4):389-407.

Takeda, H., and Y. Shimma, 1977. Effects of toxic amounts of dietary zinc on the growth and body components of rainbow trout at two levels of calcium. *Bull. Freshw. Fish. Res.* 27:103-109.

Tietge, J.E., R.D. Johnson, K.M. Jensen, P.M. Cook, G.E. Elonen, and J.D. Fernandez, 1998. Reproductive toxicity and disposition of 2,3,7,8-tetrachlorodibenzo-p-dioxin in adult brook trout (*Salvelinus fontinalis*) following a dietary exposure. *Environ. Toxicol. Chem.* 17(12):2395–2407.

Tillitt, D.E., G.T. Ankley, J.P. Giesy, J.P. Ludwig, H. Kurita-Matsuba, D.V. Weseloh, et al., 1992. Polychlorinated biphenyl residues and egg mortality in double crested cormorants from the Great Lakes. *Environ. Toxicol. Chem.* 11:1281–1288.

Tillitt, D.E., R.W. Gale, J.C. Meadows, J.L. Zajicek, P.H. Peterman, S.N. Heaton, et al., 1996. Dietary exposure of mink to carp from Saginaw Bay.3. Characterization of dietary exposure to planar halogenated hydrocarbs, dioxin equivalents, and biomagnifications. *Environ. Health Perspect.* 30(1):283–291.

T&N Associates, 2002. Dose-Response Assessment from Published Research (1981-1997) of the Toxicity of 2,3,7,8-TCDD and Other Dioxin-like Compounds to Fish and Aquatic-Associated Wildlife - Laboratory Studies. T&N Associates, Inc. Cincinnati, OH. September 2002.

USEPA, 1986. Quality Criteria for Water 1986. EPA 440/5-86-001. May 1, 1986. U.S. Environmental Protection Agency, Office of Water, Regulations and Standards, Washington, DC.

USEPA, 1988. Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1993. Wildlife Exposure Factors Handbook. Volume I of II. EPA/600/R-93/187a. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC.

USEPA, 1997. Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments. EPA 540-R-97-006. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 1999a. Screening Level Ecological Risk Assessment Protocol for Hazardous Waste Combustion Facilities. EPA530-D-99-001A. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 1999b. Phase 2 Report- Review Copy, Further Site Characterization and Analysis, Volume 2e - Baseline Ecological Risk Assessment, Hudson River PCBs Reassessment

RI/FS. Available at: http://www.epa.gov/hudson/reports.htm. Prepared by TAMS Consultants, Inc. and Menzie-Cura & Associates, Inc.

USEPA, 2000a. Bioaccumulation Testing and Interpretation for the Purpose of Sediment Quality Assessment. Status and Needs. USEPA 823-R-00-001. U.S. Environmental Protection Agency, Office of Water, and Office of Solid Waste, Washington, DC.

USEPA, 2000b. Phase 2 Report, Further Site Characterization and Analysis, Volume 2e - Revised Baseline Ecological Risk Assessment, Hudson River PCBs Reassessment. Available at: http://www.epa.gov/hudson/reports.htm. Prepared by TAMS Consultants, Inc. and Menzie-Cura & Associates, Inc.

USEPA, 2002. USEPA Region 9 Biological Technical Assistance Group (BTAG) Recommended Toxicity Reference Values for Mammals and Birds. Revision Date 11/21/2002. Available at: http://www.dtsc.ca.gov/assessingrisk/upload/eco_btag-mammal-bird-trv-table.pdf

USEPA, 2003. Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. USEPA/630/P-03/002A. U.S. Environmental Protection Agency, Risk Assessment Forum, Washington, DC.

USEPA, 2004. EPA Region 5 RCRA Ecological Screening Levels. Available at http://www.epa.gov/reg5rcra/ca/ESL.pdf. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005a. Guidance for Developing Ecological Soil Screening Levels. OSWER Directive 9285.7-55. Revised February 2005. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2005b. Ecological Soil Screening Levels for Cadmium. Interim final. OSWER Directive 9285.7-65. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005c. Ecological Soil Screening Levels for Lead. Interim final. OSWER Directive 9285.7-70. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005d. Ecological Soil Screening Levels for Vanadium. Interim Final. OSWER Directive 9285.7-75. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007a. Ecological Soil Screening Levels for Copper. Interim final. OSWER Directive 9285.7-68. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007b. Ecological Soil Screening Levels for Nickel. Interim final. OSWER Directive 9285.7-76. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2007c. Ecological Soil Screening Levels for Zinc. June 2007. OSWER Directive 9285.7-73 U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2008. Ecological Soil Screening Levels for Chromium. Interim final. OSWER Directive 9285.7-66. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response, Washington, DC.

USEPA, 2009. National Recommended Water Quality Criteria. Available at http://water.epa.gov/scitech/swguidance/standards/current/upload/nrwqc-2009.pdf. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2010. Recommended Toxicity Equivalence Factors (TEFs) for Human Health Risk Assessments of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin and Dioxin-like Compounds. EPA/100-R-10/005. Risk Assessment Forum, U.S. Environmental Protection Agency, Washington, DC. December.

USEPA, 2012. ECOTOX Database. Available at: http://cfpub.epa.gov/ecotox/. Accessed on January 2012. U.S. Environmental Protection Agency, Office of Research and Development.

Van den Berg, M., L. Birnbaum, A.T. Bosveld, B. Brunstrom, P. Cook, M. Feeley, et al., 1998. Toxic equivalency factors (TEFs) for PCBs, PCDDs, PCDFs for humans and wildlife. *Environ. Health Perspect.* 106(12):775–792.

Van den Berg, M., L.S. Birnbaum, M. Denison, M. DeVito, W. Farland, M. Feeley, H. Fiedler, H. Hakansson, A. Hanberg, L. Haws, M. Rose, S. Safe, D. Schrenk, C. Tohyama, A. Tritscher, J. Tuomisto, M. Tysklind, N. Walker, and R.E. Peterson, 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicol. Sci.* 93(2):223-241.

Villeneuve, D.C., D.L. Grant, K. Khera, D.J. Clegg, H. Baer, and W.E. Phillips, 1971. The fetotoxicity of a polychlorinated biphenyl mixture (Aroclor® 1254) in the rabbit and in the rat. *Environ. Physiol.* 1:67–71.

Voltura, M.B., and J.B. French, 2007. Effects of dietary PCB exposure on reproduction in the white-footed mouse (*Peromyscus leucopus*). *Arch. Environ. Contam. Toxicol.* 52(2):264-269.

Walker, M.I., J.L.C. Hufnagle, and R.E. Peterson, 1991a. An egg injection method for assessing early life stage mortality of polychlorinated dibenzo *p* dioxins, dibenzofurans, and biphenyls in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol.* 2:15–38.

Walker, M.K., and R.E. Peterson, 1991. Potencies of polychlorinated dibenzo-*p*-dioxin, dibenzofuran, and biphenyl congeners, relative to 2,3,7,8-tetrachlorodibenzo-*p*-dioxin, for producing early life stage mortality in rainbow trout. *Aquat. Toxicol.* 21:219–238.

Walker, M.K., and R.E. Peterson, 1994. Toxicity of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin to brook trout (*Salvelinus fontinalis*) during the early development. *Environmental Toxicology and Chemistry.* 13(5):817–820.

Walker, M.K., L.C. Hufnagle, Jr., M.K. Clayton, and R.E. Peterson, 1991a. An egg injection method for assessing early life stage mortality of polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls in rainbow trout, (*Oncorhynchus mykiss*). *Aquatic Toxicology* 22:15-38.

Walker, M.K., J.M. Spitsbergen, J.R. Olson, and R.E. Peterson, 1991b. 2,3,7,8 tetrachlorodibenzo-*p*-dioxin (TCDD) toxicity during early life stage development of lake trout (*Salvelinus namaycush*). *Can. J. Fish. Aquat. Sci.* 48:875–883.

Walker, M.K., P.M. Cook, B.C. Butterworth, E.W. Zabel, and R.E. Peterson, 1996. Potency of a complex mixture of polychlorinated dibenzo p dioxin, dibenzofuran, and biphenyl congeners compared to 2,3,7,8 tetrachlorodibenzo-*p*-dioxin in causing fish early life stage mortality. *Fundam. Appl. Toxicol* 30(2):178–186.

Walker, M.K., R.S. Pollenz, and S.M. Smith, 1997. Expression of the aryl hydrocarbon receptor (AhR) and AhR nuclear translocator during chick cardiogenesis is consistent with 2,3,7,8-tetrachlorodibenzo-p-dioxin-induced heart defects. *Toxicol. Appl. Pharmacol.* 143: 407-419.

Watling, H.R., 1983. Comparative study of the effects of metals on the settlement of *Crassostrea gigas. Bull. Environ. Contam. Toxicol. 31(3): 344-351* (as cited in USEPA 2012).

Weir, S.M., J.G. Suski, and C.J. Salice, 2010. Ecological risk of anthropogenic pollutants to reptiles: Evaluating assumptions of sensitivity and exposure. *Environmental Pollution* 158:3596-3606.

West, C.W., T. Ankley, J.W. Nichols, G.E. Elonen, and D.E. Nessa, 1997. Toxicity and bioaccumulation of 2,3,7,8-tetrachlorodibenzo-*p*-dioxin in long term tests with the freshwater benthic invertebrates *Chironomus tentans* and *Lumbriculus variegatu*s. *Environ. Toxicol. Chem.* 16(6):1287–1294.

Westin, D.T., C.E. Olney, and B.A. Rogers, 1983. Effects of parental and dietary PCBs on survival, growth, and body burdens of larval striped bass. *Bull. Environ. Contam. Toxicol.* 30:50–57.

WHO, 2001. Vanadium Pentoxide and Other Inorganic Vanadium Compounds. Concise International Chemical Assessment Document 29. ISBN 92 4 153029 4.

Williams, L.L., and J.P. Giesy, 1992. Relationships among concentrations of individual polychlorinated biphenyl (PCB) congeners, 2,3,7,8-tetrachlorodibenzo-*p*-dioxin equivalents (TCDD-EQ) and rearing mortality of chinook salmon (*Oncorhynchus tshawytscha*) eggs from Lake Michigan. *J. Great Lakes Res.* 18(1):108–124.

Willingham, E., T. Rhen, J.T. Sakata, and D. Crews, 2000. Embryonic Treatment with Xenobiotics Disrupts Steroid Hormone Profiles in Hatchling Red-Eared Slider Turtles (*Trachemys scripta elegans*). Institute of Reproductive Biology, The University of Texas, Austin, Texas, USA.

Windward, 2007. Phase 2 Remedial Investigation Report Appendix A: Baseline Ecological Risk Assessment for the Lower Duwamish Waterway Group. Seattle, WA. http://www.ldwg.org/assets/era/final_era_7_31_07.pdf. Accessed on February 6, 2012.

Windward, 2011a. Portland Harbor RI/FS: Appendix G, Baseline Ecological Risk Assessment. Draft Final. WE-09-0001. Prepared for the Lower Willamette Group. July 1, 2011. Windward Environmental LLC.

Windward, 2011b. Portland Harbor RI/FS; Appendix G, Baseline Ecological Risk Assessment, Attachment 14, Recommended Literature-Based Fish Dietary and Wildlife TRVs. Draft final. Prepared for the Lower Willamette Group. Windward Environmental LLC.

Wintermyer, M.L., and K.R. Cooper, 2003. Dioxin/furan and polychlorinated biphenyl concentrations in eastern oyster (*Crassostrea virginica*, Gmelin) tissues and the effects on egg fertilization and development. *Journal of Shellfish Research* 22(3):737-746.

Wintermyer, M.L., and K.R. Cooper, 2007. The development of an aquatic bivalve model: evaluating the toxic effects on gametogenesis following 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) exposure in the eastern oyster (*Crassostrea virginica*). *Aquatic Toxicology* (Amsterdam) 81(1):10-26.

Wintermyer, M., A. Skaidas, A. Roy, Y. Yang, P. Georgapoulos, J. Burger, and K.R. Cooper, 2005. The Development of a Physiologically-Based Pharmacokinetic (PBPK) Model Using the Distribution of 2,3,7,8-Tetrachlorodibenzo-*p*-Dioxin (2,3,7,8-TCDD) in the Tissues of the Eastern Oyster (*Crassostrea virginica*). *Mar. Environ. Res*. 60:133-152.

Wobeser, G., N.O. Nielsen, and B. Schiefer, 1976. Mercury and mink. II. Experimental methyl mercury intoxication. *Can. J. Comp. Med.* 40(1):34–45.

Woodford, J.E., W.H. Karasov, M.W. Meyer, and L. Chambers, 1998. Impact of 2,3,7,8-TCDD Exposure on Survival, Growth, and Behavior of Ospreys Breeding in Wisconsin, USA. *Environmental Toxicology and Chemistry* 17(7):1323-1331.

Wren, C.D., 1991. Cause effect linkages between chemicals and populations of mink (*Mustela vison*) and otter (*Lutra canadensis*) in the Great Lakes Basin. *J. Toxicol. Environ. Health* 33(4):549–585.

Yockim, R.S., A.R. Isensee, and G.E. Jones, 1978. Distribution and toxicity of TCDD and 2,4,5-T in an aquatic model ecosystem. *Chemosphere* 3:215-220.

Zabel, E.W., P.M. Cook, and R.E. Peterson, 1995. Toxic equivalency factors of polychlorinated dibenzo p dioxin, dibenzofuran and biphenyl congeners based on early life stage mortality in rainbow trout (*Oncorhynchus mykiss*). *Aquat. Toxicol.* (Amsterdam) 31(4):315–328.

Zwiernik, M.J., K.J. Beckett, S. Bursian, D.P. Kay, R.R. Holem, J.N. Moore, B. Yamini, and J.P. Giesy. 2009. Chronic effects of polychlorinated dibenzofurans on mink in laboratory and field environments. *Integr Environ Assess Management* 5(2):291-301.

TABLES

Table B-1
Chemicals of Potential Ecological Concern[a]

Chemical	Receptors North of I-10 and Aquatic Environment	
	Benthic Invertebrates	Fish and Wildlife
Dioxins/Furans		
Dioxins and Furans	X	X
Polychlorinated Biphenyls		
Polychlorinated Biphenyls		X
Semivolatile Organic Compounds		
Bis(2-ethylhexyl)phthalate	X	X
Carbazole	X	
Phenol	X	
Metals		
Aluminum	X	
Barium	X	
Cadmium		X
Chromium		
Cobalt	X	
Copper	X	X
Lead	X	
Manganese	X	
Mercury	X	X
Nickel		X
Thallium	X	
Vanadium	X	
Zinc	X	X

Notes
a - Some of these chemicals will be evaluated for human health risks. See Integral (2012).

Table B-2

Default Values for Body Weight and Consumption Rates for Test Species Used in Derivation of TRVs

Test Species	Body Weight (kg)	Consumption Rate (kg dw/day)
Mammals		
Mink	1.0 [a]	0.137 [a]
Mouse	0.03 [a]	NA
Rat	0.35 [a]	0.028 [a]
Birds		
Japanese quail	0.15 [a]	0.0169 [a]
Mallard duck	1.13 [b]	NA
Starling	NA	NA
Turkey	NA	0.174 [a]

Notes

NA = not available

a - Sample et al. (1996)

b - USEPA (1993)

Table B-3
Ambient Water Quality Criteria for the Protection of Aquatic Life

Chemical	Water Concentration[a] (µg/L)	
Organic Compounds		
PCBs[b]	CMC	NA
	CCC	0.03
Metals		
Cadmium[b]	CMC	40
	CCC	8.8
Copper	CMC	4.8
	CCC	3.1
Lead	CMC	210
	CCC	8.1
Mercury	CMC	1.8
	CCC	0.94
Nickel	CMC	74
	CCC	8.2
Zinc	CMC	90
	CCC	81

Notes

CCC = criterion continuous concentration: an estimate of the highest concentration in ambient water to which an aquatic community can be exposed indefinitely without an adverse effect.

CMC = criterion maximum concentration: an estimate of the highest concentration in ambient water to which an aquatic community can be exposed briefly without an adverse effect.

NA = not available

PCB = polychlorinated biphenyl

a - National ambient water quality criteria (AWQC) for the protection of aquatic life. Values for saltwater are shown. These AWQC values represent the dissolved concentration.

b - Chemical is a chemical of potential ecological concern for fish only.

Table B-4
Summary of Information on the Toxicity of 2,3,7,8-TCDD[a] to Benthic Invertebrates

Exposure Medium	Test Organism	Taxonomic Classification	Dose Administration	Exposure Duration	NOAEC/LOAEC	Units	Endpoint	Notes	Reference
Sediment	*Ampelisca abdita*	Crustacea, Amphipoda	Spiked sediment	10 days	25,000/NA	ng/kg dw sediment	Growth and mortality		Barber et al. (1998)
	Nereis virens	Annelida, Polychaeta	Field-collected sediment	180 days	656/NA	ng/kg dw sediment	Mortality	Potential co-contamination with 2,3,7,8-TCDF and PCBs noted.	Pruell et al. (1993)
					422/NA	ng/kg dw tissue			
	Macoma nasuta	Mollusca, Bivalvia	Field-collected sediment	120 days	656/NA	ng/kg dw sediment	Mortality	Presence of 2,3,7,8-TCDF and PCBs in both contaminated sediments and in study organisms noted.	Rubenstein et al. (1990)
					142/NA	ng/kg dw tissue			
	Palaemonetes pugio	Crustacea, Caridea	Field-collected sediment	28 days	656/NA	ng/kg dw sediment	Mortality		
					138/NA	ng/kg dw tissue			
	Chironomus riparius	Arthropoda, Diptera	Spiked sediment	28 days	10,000/NA	ng/kg dw sediment	Mortality, growth, mentum deformities		Loonen et al. (1996)
					14,000/NA	ng/kg dw tissue			
Water	*Daphnia magna*	Crustacea, Cladocera	Laboratory water	48 hours followed by 7 day recovery	1,030/NA	ng/kg ww tissue[b]	General toxicity		Adams et al. (1986)
	Mya arenaria	Mollusca, Bivalvia	Laboratory water	Single pulse dose for 24 hours followed by 28 day observation period	200/NA	ng/L in water	Reduced body mass over time		Cooper and Wintermyer (2009)
					NA/4.8 - 20	ng/kg ww weight tissue[b]	Gonadal lesions (female)		
	Physa sp.	Mollusca, Gastropoda	Well water	36 days followed by recovery period	200/NA	ng/L in water	Parental mortality, hatching, juvenile mortality		Miller et al. (1973)
	Paranais sp.	Annelida, Oligochaeta	Well water	55 days	200/NA	ng/L in water	Total biomass		
	Aedes aegypti	Arthropoda, Diptera	Well water	17 days followed by recovery period	200/NA	ng/L in water	Pupation		
	Mya arenaria	Mollusca, Bivalvia	Sea water	24 hours followed by recovery period	2,000/NA	ng/L in water	Mortality, shell length, gonadal histopathology	Tissue concentrations were measured but were widely variable among organs.	Rhodes et al. (1997)
	Helisoma sp.	Mollusca, Gastropoda	Spiked soil flooded with water	32 days	4.2/NA	ng/L in water	Reproductive activity, feeding, growth		Yockim et al. (1978)
	Daphnia magna	Crustacea, Cladocera	Spiked soil flooded with water	32 days	4.2/NA	ng/L in water	Reproductive activity, feeding, growth		
	Physa sp.	Mollusca, Gastropoda	Water		1,300/NA	ng/L in water	Reproductive activity, growth, feeding		Isensee and Jones (1975)
	Daphnia magna	Crustacea, Cladocera	Water		1,300/NA	ng/L in water	Reproductive activity, growth, feeding		

Table B-4
Summary of Information on the Toxicity of 2,3,7,8-TCDD[a] to Benthic Invertebrates

Exposure Medium	Test Organism	Taxonomic Classification	Dose Administration	Exposure Duration	NOAEC/LOAEC	Units	Endpoint	Notes	Reference
Diet	*Chironomus dilutus*	Arthropoda, Diptera	Spiked diet	35 days	3,804/NA	µg/kg TOC diet	Mortality, growth emergence, eggs/female, hatchability	TCDD concentrations also given as dw of food (323 µg/kg dw diet) and also provided on a lipid basis in *Chironomus* , see below.	West et al. (1997)
	Lumbriculus variegatus	Annelida, Oligochaeta	Spiked diet	28 days	3,594/NA	µg/kg TOC diet	Number of organisms, total biomass	Trend for reduced number of animals, but not statistically significant. TCDD concentration also given as dw of food (1,319 µg/kg dw diet) and also provided on a lipid basis in *Lumbriculus* tissue, see below.	
	Chironomus dilutus	Arthropoda, Diptera	Spiked diet	35 days	5,084	µg/kg lipid	Mortality, growth emergence, eggs/female, hatchability	Highest concentration (average of exposure group) during exposure period, achieved at day 13.	West et al. (1997)
	Lumbriculus variegatus	Annelida, Oligochaeta	Spiked diet	28 days	9,533/NA	µg/kg lipid	Number of organisms, total biomass	Highest concentration (average of exposure group), achieved at end of exposure period.	
Administered/ Injection	*Mya arenaria*	Mollusca, Bivalvia	Injection (muscle; single dose)		200/NA	ng/kg ww tissue[b]	Reduced body mass over time		Cooper and Wintermyer (2009)
			Siphon gavage (single dose)		NA/200	ng/kg ww tissue[b]	Reduced body mass over time		
	Crassostrea virginica	Mollusca, Bivalvia	Injection (Days 1 and 14)	28 day observation period	NA/2.0	ng/kg ww tissueb[a]	Reduced body mass over time		Cooper and Wintermyer (2009)
					NA/2.0	ng/kg ww tissue[b]	Gonadal lesions		
					NA/2	ng/kg ww tissue[b]	Reduced larval survival		
	Crassostrea virginica	Mollusca, Bivalvia	Injection (Days 1 and 14)	28 day observation period	NA/2	ng/kg ww tissue[b]	Delayed gonadogenesis (females)	Marked effect of solvent on this endpoint.	Wintermyer and Cooper (2007)
					NA/10	ng/kg ww tissue[b]	Sex ratio (reduced females)		
					NA/2	ng/kg ww tissue[b]	Reduced vitellogenic oocytes (females; electron microscopy)	Other reproductive endpoints also affected this exposure level.	
					2/10	ng/kg ww tissue[b]	Delayed gonadogenesis (males)		

Notes

Animals exposed to field-collected sediment may have been exposed to mixtures.
LOAEC = lowest-observed-adverse-effects concentration
NA = not available
NOAEC = no-observed-adverse effects concentration
PCB = polychlorinated biphenyl
TCDD = tetrachlorodibenzo-*p* -dioxin
TOC = total organic carbon
a - All laboratory studies summarized here used 2,3,7,8-TCDD as the exposure chemical. Some field studies summarized also measured other organocontaminants and these have been summarized in the Notes column.
b - Soft body tissue only (excluding shell)

Table B-5

Geometric Means of No-Observed-Effect Levels and Lowest-Observed-Effect Levels of 2,3,7,8-Tetrachlorodibenzo-*p* -Dioxin (2,3,7,8-TCDD) and Dioxin-Like Compounds in Fish Eggs or Embryos

Common Name	Scientific Name	GM [a] (ng/g lipid)	Reference
Brook trout	*Salvelinus fontinalis*	1.68	b
Channel catfish	*Ictalurus punctatus*	11.95	c
Fathead minnow	*Pimephales promelas*	13.32	c
Japanese medaka	*Oryzias latipes*	22.66	c
Lake herring	*Coregonus artedii*	3.29	c
Lake trout	*S. namaycush*	0.42	d
Northern pike	*Esox lucius*	34.85	c
Rainbow trout	*O. mykiss*	3.05	e
White sucker	*Catastomus commersoni*	40.69	c
Zebra fish	*Danio danio*	54.17	c

Notes

GM = geometric mean

a - Values are listed with the number of significant figures presented in the respective study.

b - Johnson et al. (1998), as cited in Steevens et al. (2005)

c - Elonen et al. (1998), as cited in Steevens et al. (2005)

d - Walker et al. (1994), as cited in Steevens et al. (2005)

e - Walker and Peterson (1991), as cited in Steevens et al. (2005)

Table B-6
Summary of Egg Mortality TRVs; Maternal Transfer and Yolk Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	328	1,477	MT	a	Egg concentrations estimated on the basis of maternal dose of 1 µg/kg for no effects and an estimated 50 percent egg mortality at 4.5 µg/kg bw, assuming a 1 percent maternal transfer into eggs (mean egg wt = 30.5 g) Nosek et al. (1992a; 1993).
$[Egg]_{TCDD}$	100	1,000	YI	b	Egg concentration associated with 10 percent egg mortality
Geometric mean for pheasants	181	1,215			
Double crested cormorant					
$[Egg]_{TCDD}$	1,000	4,000	YI	c	LOAEL is associated with 23.3 percent increase in egg mortality over egg mortality in vehicle controls
$[Egg]_{TCDD}$	1,300	5,400	YI	d	LOAEL is associated with 25.5 percent increase in egg mortality over egg mortality in vehicle controls
Geometric mean for cormorants	1,140	4,648			
Final geometric mean	450	2,400			Geometric means rounded to 2 significant figures for use as TRVs
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	YI	e	LOAEL is associated with 100 percent egg mortality over control egg mortality
$[Egg]_{TCDD}$	80	160	YI	f	LOAEL is associated with 63.8 percent increase in egg mortality over egg mortality in vehicle controls
Geometric mean for chickens	89	220			
Geometric mean - all	260	1,100			

Notes

LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
MT = maternal transfer
NOAEC = no-observed-adverse effects concentration
TRV = toxicity reference value
YI = yolk injection

a - Nosek et al. (1992b)
b - Nosek et al. (1993)
c - Powell et al (1997a)
d - Powell et al. (1998)
e - Henshel et al. (1997a)
f - Powell et al. (1996)

Table B-7
Toxicity Equivalency Factors for Dioxins and Furans and Dioxin-Like PCBs

Compound	TEF-M (WHO 2005) [a]	TEF-Fish (WHO 1998)	TEF-Bird (WHO 1998)
Chlorinated Dibenzo-*p* -Dioxins			
2,3,7,8-TCDD	1	1	1
1,2,3,7,8-PeCDD	1	1	1
1,2,3,4,7,8-HxCDD	0.1	0.5	0.05
1,2,3,6,7,8-HxCDD	0.1	0.01	0.01
1,2,3,7,8,9-HxCDD	0.1	0.01	0.1
1,2,3,4,6,7,8-HpCDD	0.01	0.001	0.001
OCDD	0.0003	0.0001	0.0001
Chlorinated Dibenzofurans			
2,3,7,8-TCDF	0.1	0.05	1
1,2,3,7,8-PeCDF	0.03	0.05	0.1
2,3,4,7,8-PeCDF	0.3	0.5	1
1,2,3,4,7,8-HxCDF	0.1	0.1	0.1
1,2,3,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,7,8,9-HxCDF	0.1	0.1	0.1
2,3,4,6,7,8-HxCDF	0.1	0.1	0.1
1,2,3,4,6,7,8-HpCDF	0.01	0.01	0.01
1,2,3,4,7,8,9-HpCDF	0.01	0.01	0.01
OCDF	0.0003	0.0001	0.0001
Non-ortho Substituted PCBs			
3,3',4,4'-Tetrachlorobiphenyl (PCB 77)	0.0001	0.0001	0.05
3,4,4',5-Tetrachlorobiphenyl (PCB 81)	0.0003	0.0005	0.1
3,3',4,4',5-Pentachlorobiphenyl (PCB 126)	0.1	0.005	0.1
3,3',4,4',5,5'-Hexachlorobiphenyl (PCB 169)	0.03	0.00005	0.001
Mono-ortho Substituted PCBs			
2,3,3',4,4'-Pentachlorobiphenyl (PCB 105)	0.00003	0.000005	0.0001
2,3,4,4',5-Pentachlorobiphenyl (PCB 114)	0.00003	0.000005	0.0001
2,3',4,4',5-Pentachlorobiphenyl (PCB 118)	0.00003	0.000005	0.00001
2',3,4,4',5-Pentachlorobiphenyl (PCB 123)	0.00003	0.000005	0.00001
2,3,3',4,4',5-Hexachlorobiphenyl (PCB 156)	0.00003	0.000005	0.0001
2,3,3',4,4',5'-Hexachlorobiphenyl (PCB 157)	0.00003	0.000005	0.0001
2,3',4,4',5,5'-Hexachlorobiphenyl (PCB 167)	0.00003	0.000005	0.00001
2,3,3',4,4',5,5'-Heptachlorobiphenyl (PCB 189)	0.00003	0.000005	0.00001

Sources

WHO (1998) corresponds to Van den Berg et al. (1998)

WHO (2005) corresponds to Van den Berg et al. (2006)

Notes

PCB = polychlorinated biphenyl
TEF-M = mammalian toxicity equivalency factor
a - Endorsed by USEPA (2010)

Table B-8
Summary of Egg Mortality TRVs for TCDD; Air Cell or Albumin Injection Studies

Exposure Parameter	NOAEC ng/kg ww	LOAEC ng/kg ww	Egg Exposure	Ref	Comments
Japanese quail					
$[Egg]_{TCDD}$	3,540	9,020	CI	a	Egg mortality significantly elevated over control
Ring-necked (or common) pheasant					
$[Egg]_{TCDD}$	264	1,030	CI	a	Egg mortality significantly elevated over control
$[Egg]_{TCDD}$	100	1,000	AI	b	LOAEL is egg concentration associated with 20 percent mortality over control
Domestic Chicken					
$[Egg]_{TCDD}$	100	300	CI	c	LOAEL is egg concentration associated with 35 percent increase in mortality over control
$[Egg]_{TCDD}$	248	515	CI	a	Egg mortality significantly elevated over control

Notes

AI = albumin injection
CI = air cell injection
LOAEC = lowest-observed-adverse-effects concentration
LOAEL = lowest observed adverse effect level
NOAEC = no-observed-adverse effects concentration
TCDD = tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
a - Cohen-Barnhouse et al. (2011)
b - Nosek et al. (1993)
c - Henshel et al. (1997a)

Table B-9
Summary of NOAECs for Bird Eggs, Field Studies

Receptors	TEQ (ng/kg ww)	Location	Ref.	Comments
Spotted sandpipers	732	Hudson River, NY	Custer and Custer (2010)	NOAEC is based on the geometric mean of TEQ concentrations measured for the Hudson River site
Great blue heron	207	British Columbia, Canada	Elliott et al. (2001)	Based on lack of gross abnormalities, number of fledglings and hatching success at the Nicomekl River site.
Cormorants, herons and egrets	452	Galveston Bay, Tx	Frank et al. (2001)	TEQ ranged from 67 to 452 ng/kg ww, no deformities or abnormalities were detected over this range.
Osprey	136	Castle Rock and Petenwell Flowages site, WI	Woodford et al. (1998)	Using the reproductive endpoints of egg hatching and chick fledgling rates

Notes

NOAEC = no-observed-adverse effects concentration

Table B-10
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	0.98	a	Reproduction	Geometric means of NOAELs and LOAELs from toxicity studies with mice.
	LOAEL	2			
TCDD	NOAEL	0.000001	b	Reproduction	Converted from dietary concentration to dose using assumed body weight and consumption rate.
	LOAEL	0.00001			
Bis(2-ethylhexyl)phthalate	NOAEL	5.8	c	Reproduction	Effects seen at 29 and 147 mg/kg/day doses might be age-related, in which case NOAEL and LOAEL would be under-estimated
	LOAEL	29			
Metals					
Cadmium	NOAEL	2	d	Geometric mean of bounded NOAELs for growth, mortality, repro	38 bounded NOAELs/LOAELs included in calculation
	LOAEL	10		Geometric mean of associated LOAELs	
Chromium	NOAEL	2.40	e	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.82		Mortality	No unbounded LOAELs. This is the minimum unbounded LOAEL for a mortality/growth/repro endpoint.
Copper	NOAEL	5.6	f	Reproduction, growth, survival	Highest bounded NOAEL beneath the lowest bounded LOAEL
	LOAEL	9.34			
Lead	NOAEL	4.7	g	Survival	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	5.0		Growth	Lowest bounded LOAEL
Mercury	NOAEL	0.015	h	Survival and growth	Converted from dietary concentration to dose using assumed body weight and consumption rate. Converted to chronic from subchronic exposure period. Administered as methylmercury chloride.
	LOAEL	0.025			
Nickel	NOAEL	1.7	i	Reproduction	Highest bounded NOAEL below the lowest bounded LOAEL for a mortality/growth/repro endpoint
	LOAEL	2.71			Minimum bounded LOAEL for a mortality/growth/repro endpoint

Table B-10
Toxicity Reference Values for Mammals

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Zinc	NOAEL	75.4	j	Reproduction	Geometric mean of NOAELs for reproduction and growth; lowest bounded LOAEL for survival, reproduction and growth
	LOAEL	75.9			

Notes
EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/

LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Aulerich and Ringer (1977)
b - Murray et al. (1979)
c - David et al. (2000)
d - EcoSSL (USEPA 2005b)
e - EcoSSL (USEPA 2008)
f - EcoSSL (USEPA 2007a)
g - EcoSSL (USEPA 2005c)
h - Sample et al. (1996)
j - USEPA (2007c)

Table B-11
Summary of Toxicity Reference Values for Total PCB Effects on Fish, Birds, and Mammals

Receptors	TRV_{NOAEL} GM (range)	TRV_{LOAEL} GM (range)	Units	TRV Basis	Sample Size
Fish	5.0 (1.9–15)	16 (2.7–170)	mg/kg ww	Residue in whole fish	NOAEL n=3 across 3 fish species; LOAEL n=3 across 3 fish species
Birds	≥2 (0.1–7)	≤3 (1–10)	mg/kg-day	Ingested dose	NOAEL n=9 across 5 bird species; LOAEL n=7 across 4 bird species
Mink	≥0.2 (0.1–0.1)	≤0.2 (0.7–0.7)	mg/kg-day	Ingested dose	Mink toxicity studies; NOAEL n=4; LOAEL n=7
Other Mammals	0.98 (0.36–2.6)	≤2 (4–6)	mg/kg-day	Ingested dose	Mouse toxicity studies; NOAEL n=2; LOAEL n=5

Notes

GM = geometric mean. A “>” or “<” sign is used where one or more of the values used to determine the geometric mean TRV was an unbounded NOAEL or LOAEL, respectively. Range is the range of bounded TRVs (i.e., across accepted studies in which both a NOAEL and a LOAEL were reported). Sample size is all bounded and unbounded TRVs from accepted studies before combining studies within species.

LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
TRV = toxicity reference value
ww = wet weight

Table B-12
Data Sources Used to Develop Fish Tissue-Based Toxicity Reference Values for Total PCBs

Chemical	Chemical Form	Test Organism	Study Duration	Endpoints	Exposure Route	Dosage/Exposure	Reported Toxicity	Test Species NOAEL	Test Species LOAEL	TRV_{NOAEL} (mg/kg)	TRV_{LOAEL} (mg/kg)	Value Used for ERA	Source Category	Source/Comments
Total PCBs	Aroclor-1254	Sheephead Minnow	28 days	F1 Generation Survival	Aqueous	Five water conc (0.1, 0.32, 1.0, 3.2, and 10 µg/L) plus control	NOAEL, LOAEL	0.1 µg/L (1.9 mg/kg in adult)	0.32 µg/L (9.3 mg/kg in adult)	1.9	9.3	Yes	Lit	Hansen et al. (1973); NOAEL and LOAEL were used for the Onondaga Lake BERA (NYSDEC 2002) and Hudson River Revised BERA (USEPA 2000)
Total PCBs	Clophen A50	Common Minnow	40 days exposed, monitor for additional 260 days	Growth, F1 Generation Survival	Diet	Control and 3 dietary conc (20, 200 and 2,000 mg/kg)	NOAEL, LOAEL	15 mg/kg	170 mg/kg	15	170	Yes	Lit	Bengtsson (1980)
Total PCBs	20 PCBS (representing the 154 tetra- to hepta-chlorinated congeners)	Zebrafish	13 weeks exposed; reproduction study initiated at 9 weeks	Growth, Survival, Reproduction	Diet	Control and 3 dietary conc (0.008, 0.08, and 0.4 mg/kg)	LOAEL	NR	2.7 mg/kg	---	2.7	Yes	Lit	Orn et al. (1998). Fish were dissected and livers and ovaries removed prior to measuring concentrations. LOAEL value therefore is biased low.
Total PCBs	Aroclor-1016, Aroclor-1254, Aroclor-1260	Striped Bass	30 days (10 days yolk absorption, 20 days exposed)	Growth, Survival	Diet	Control and 1 dietary conc (0.014 and 0.127 mg/kg, respectively)	NOAEL	3.1 mg/kg	NR	4.4	---	Yes	Lit	Westin et al. (1983). No effects observed in dosed fish. Highest concentration in whole larvae selected as TRV.

Notes

TRV values are on a whole body wet weight basis.

When multiple NOAEL or LOAEL values are reported based on the same endpoint, the highest reported value was used for the TRV calculations.

Study durations shown were as reported by the authors and were also adjusted to days to facilitate comparisons between studies. Where an unbounded LOAEL was greater than the maximum bounded LOAEL, it was excluded from calculation of the geometric mean LOAEL.

Where an unbounded NOAEL was less than the minimum bounded NOAEL, it was excluded from calculation of the geometric mean NOAEL.

BERA = baseline ecological risk assessment
ERA = ecological risk assessment
LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
NR = not reported or not required
PCB = polychlorinated biphenyl
TRV = toxicity reference value
USACE = U.S. Army Corps of Engineers

Table B-13
Data Sources Used to Develop Total PCBs Toxicity Reference Values for Mice

Chemical	Chemical Form	Study Duration	Endpoints	Exposure Route	Dosage	Reported Toxicity Value(s)	NOAEL	LOAEL	Calculated TRV_{NOAEL} (mg/kg-day)	Calculated TRV_{LOAEL} (mg/kg-day)	Source/Comments
Total PCBs	Aroclor 1254	9 to 18 months (270 to 540 days)	Repro	Oral in diet	10 ppm in diet	LOAEL	---	10 mg/kg	---	1.27	Linzey (1987)
Total PCBs	Aroclor 1254	F1 generation 4 to 12 weeks (28 to 84 days)	Repro, Survival	Oral in diet	10 ppm in diet	LOAEL	---	10 mg/kg	---	1.27	Linzey (1988) [a]
Total PCBs	Aroclor 1254	21 days	Survival	Oral in diet	Four dose levels: 2.5, 25, 50 and 100 ppm	NOAEL, LOAEL	2.5 ppm	25 ppm in diet	0.36	3.6	Simmons and McKee (1992) TRV_{NOAEL} value reported by USEPA (2002).
Total PCBs	Aroclor 1254	3 generations (1 year) (365 days)	Repro	Oral in diet	5 ppm in diet	LOAEL	---	5 mg/kg	---	0.68	McCoy et al (1995)
Total PCBs	Aroclors 1242 and 1254	4 months (120 days)	Growth, Repro,	Oral in diet	Two dose levels: 10 and 25 ppm as total PCBs	NOAEL, LOAEL	10 ppm in diet	25 ppm in diet	2.64	6.19	Voltura and French (2007) PCB mixture was 2:1 Aroclor 1242:Aroclor 1254.
							Accepted Studies	*Range*	0.36 - 2.64	0.68 - 6.19	
								Range of bounded values	0.36 - 2.64	3.60 - 6.19	
								Geometric Mean	0.98	2	

Notes

All mouse toxicity studies shown in this table were accepted for calculation of TRVs.
The geometric means of the TRVs were body-weight scaled to derive TRVs for mammals other than mink. TRVs specific for mink were available.
Study durations shown were as reported by the authors and were also adjusted to days to facilitate comparisons between studies.

LOAEL = lowest-observed-adverse-effect level
NOAEL = no-observed-adverse-effect level
PCB = polychlorinated biphenyl
TRV = toxicity reference value

a - FI generation were offspring of parents from prior study (Linzey 1987).

Table B-14
Toxicity Reference Values and Benchmarks for Benthic Macroinvertebrates

Chemical	Sediment Concentration (ng/kg dw for organics; mg/kg dw for metals)		Ref	Water Concentration[a] (µg/L)		Ref	Endpoint/Comments
	TRV Type	Value		TRV Type	Value		
Organic Compounds							
2,3,7,8-TCDD	NOAEC	2,343		NA	NA		Geometric mean of NOAECs for a range of invertebrate taxa from Table B-4
Bis(2-ethylhexyl)phthalate	--	ND		NOAEC[b]	100	c	Opossum shrimp and amphipod mortality in 4 day lab test. NOAEC is LC_{50} ÷10.
Carbazole	--	ND					No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Phenol	--	ND		NOAEC[b]	26	d	Mysid shrimp mortality in 4 day lab test. NOAEC is LC_{50} ÷ 10.
Metals							
Aluminum	--	ND		NOAEC[b]	1,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Barium	--	ND		--	ND		No marine invertebrate data were available in ECOTOX. No sediment or water TRVs were found in the literature.
Cobalt	--	ND		NOAEC[b]	450	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Copper	ER-L	34	f	--			
	ER-M	270	f	AWQC (CCC)	3.1	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]
Lead	ER-L	46.7	f	--			
	ER-M	218	f				
Manganese	--	ND		NOAEC[b]	7,000	e	Derived from 96-hour LC_{50} with Harpacticoid copepod. NOAEC is LC_{50} ÷ 10.
Mercury	ER-L	0.15	f	--			
	ER-M	0.71	f	AWQC (CCC)	0.94	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]
Thallium	--	ND		NOAEC[b]	213	h	Derived from acute toxicity to marine life . NOAEC is EC ÷ 10. Details unavailable.
Vanadium	--	ND		NOAEC	5	i	NOAEC is EC_{50} ÷10 in most sensitive species. Effect is development.
				LOAEC	10	i	LOAEC is EC_{50} ÷ 10 in most sensitive species. Effect is development.
Zinc	ER-L	150	f	--			
	ER-M	410	f	AWQC (CCC)	81	g	AWQC (CCC) values are concentrations at or below which unacceptable effects are not expected.[g]

Notes

-- = Risks were not evaluated using lines of evidence requiring this information
AWQC = Ambient Water Quality Criteria. Criterion Continuous Concentrations shown
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
EC = effects concentration
ER-L = effect range-low: concentration below which effects are rarely observed or predicted among sensitive life stages and (or) species of biota
ER-M = effect range-median: concentration above which effects are frequently or always observed among most species of biota
USEPA = U.S. Environmental Protection Agency
WHO = World Health Organization
a - TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - TRV is an LC_{50} divided by an uncertainty factor of 10.
c - Ho et al. (1997)
d - Kim and Chin (1995)
e - Bengtsson (1978)
f - Long et al. (1995)
g - Ambient Water Quality Criteria Website
(http://water.epa.gov/scitech/swguidance/standards/current/index.cfm#altable)
h - USEPA (1986)
i - WHO (2001)

Table B-15
Toxicity Reference Values and Benchmarks for Fish

Chemical	Water Concentration[a] µg/L		Ref	Fish Food[b] (mg/kg dw)		Ref	Fish Whole Body		Units	Ref	Comments
Organic Compounds											
TCDD (mg/kg lipid)	--	--		--	--		NOAEL	0.321	µg/kg lipid	c	From a species sensitivity distribution; protects 95 percent of fish species. Endpoint is egg survival.
PCBs	--	--		--	--		NOAEL	5.0	mg/kg ww	d	Geometric mean of NOAELs from 3 fish species.
	--	--		--	--		LOAEL	16	mg/kg ww	d	Geometric mean of LOAELs across 3 fish species.
Bis(2-ethylhexyl)phthalate	NOAEL	55,000	e	--	--		--	--	--		Derived from 4-day acute test with sheepshead minnow. NOAEL is LC_{50} ÷ 10. Endpoint is survival.
Metals											
Cadmium				LOAEL	14.1	f	--	--	--		
Copper				NOAEL	50	g	--	--	--		
				LOAEL	100	h	--	--	--		
Mercury				NOAEL	0.5	i	--	--	--		Endpoint is F_0 male survival in mummichog resulting from increased aggression due to neurotoxic effects. aquarium confinement, or both.
				LOAEL	1.9	i	--	--	--		
Nickel	NOAEL	3,600	j, k	ND			--	--	--		Geometric mean of NOAECs for several marine fish. See Table B-16 and Appendix B text.
Zinc				NOAEL	1,900	l	--	--	--		Fish exposed to multiple metals in water as well as food. Fish fed live Artemia exposed to zinc chloride in water. Endpoints are growth and survival.
				LOAEL	2,000		--	--	--		Fish fed at same dose of zinc with 0.5% calcium experienced no adverse effects. Endpoint is growth.

Notes

AWQC = ambient water quality criteria
CCC = Criterion Continuous Concentration
CMC = Criterion Maximum Concentration
LOAEL = lowest observed adverse effect level
NOAEL = no observed adverse effect level
TRV = toxicity reference value
-- = Risks were not evaluated using lines of evidence requiring this information.

a - Includes AWQC and TRVs as concentrations in water for those chemicals with no AWQC (see Table B-3)
b - Windward (2011). Values presented are lowest NOAEC with a bounded LOAEC.
c - Steevens et al. (2005)
d - See Table B-11
e - TRV is an LC_{50} divided by an uncertainty factor of 10
f - Hatakayama and Yasuo (1987), as cited in Windward (2011b)
g - Windward (2011b)
h - Windward (2011b)
i - Matta et al. (2001)
j - Hunt et al. (2002)
k - USEPA (1988) Ambient Water Quality Criteria Document for Nickel
l - Windward (2007)

Table B-16
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Organic Compounds					
PCBs	NOAEL	2	a	Reproduction	Geometric mean of NOAELs for 5 bird species (Table B-11)
	LOAEL	3			Geometric mean of LOAELs for 4 bird species (Table B-11)
TCDD (ingested dose)	NOAEL ng/kg-d	14	b	Hen mortality and egg mortality	Ingested dose was estimated from weekly injected dose.
	LOAEL ng/kg-d	140			
TCDD (egg concentration ng/kg ww)	NOAEL	450	c	Egg mortality	Derived from multiple studies. See Appendix B
	LOAEL	2,400			
Bis(2-ethylhexyl)phthalate	NOAEL	74.9	d	Growth	Unbounded NOAEL for body weight
	LOAEL	--			
Metals					
Cadmium	NOAEL	1.47	e	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.37		Reproduction	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Chromium	NOAEL	2.66	f	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	2.78			Minimum bounded LOAEL for a mortality/growth/repro endpoint
Copper	NOAEL	4.05	g	Reproduction, growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	12.1			
Lead	NOAEL	1.63	h	Reproduction	Highest bounded NOAEL below lowest bounded LOAEL
	LOAEL	1.94			Lowest bounded LOAEL
Mercury	NOAEL	0.078	i	Reproduction	One dose only tested. Unbounded NOAEL for first generation.
	LOAEL	0.9	j	Reproduction	Administered as methylmercury.

Table B-16
Toxicity Reference Values for Birds

Chemical		TRV (mg/kg bw-day)	Ref	Endpoint	Comments
Nickel	NOAEL	6.71	k	Reproduction, growth	Geometric mean of NOAELs for reproduction and growth
	LOAEL	11.5		Growth	Minimum bounded LOAEL for a mortality/growth/repro endpoint
Vanadium	NOAEL	0.344	l	Growth	Highest bounded NOAEL below the lowest bounded LOAEL for survival, growth, or reproduction
	LOAEL	0.413		Reproduction	Lowest bounded LOAEL for survival, growth, or reproduction
Zinc	NOAEL	66.1	m	Reproduction	Geometric mean of NOAELs for reproduction and growth
	LOAEL	86.6			Lowest bounded LOAEL for survival, growth, or reproduction

Notes

EcoSSL = Interim EcoSSL Documents by chemical. Available at: http://www.epa.gov/ecotox/ecossl/
LOAEL = lowest observed adverse effect level
NA = not available
NOAEL = no observed adverse effect level
PCB = polychlorinated biphenyl
TCDD = 2,3,7,8-tetrachlorodibenzo-*p* -dioxin
TRV = toxicity reference value
USEPA = U.S. Environmental Protection Agency

a - Risebrough and Anderson (1975)
b - Nosek et al. (1992a)
c - Appendix B
d - O'Shea and Stafford (1980)
e - EcoSSL (USEPA 2005b)
f - EcoSSL for Cr(III) (USEPA 2008)
g - EcoSSL (USEPA 2007a)
h - EcoSSL (USEPA 2005c)
i - Heinz (1979)
j -Hill and Schaffner (1976)
k - EcoSSL (USEPA 2007b)
l - EcoSSL (USEPA 2005d)
m - USEPA (2007c)

Table B-17
Data Used to Derive Nickel TRV for Fish

Common Name	Latin Name	SMAV (µg/L)[a]	Estimated NOEC	NOEC[b]
Mummichog (adult)	*Fundulus heteroclitus*	149,900	14,990	
Atlantic silverside (larva)	*Menldia menldia*	7,960	796	
Tidewater silverside (juvenile)	*Menldia peninsulae*	38,000	3,800	
Striped bass	*Moreone saxatilis*	21,000	2,100	
Spot (juvenile)	*Lelostomus xanthurus*	70,000	7,000	
Topsmelt	*Atherinops affinis*	26,560		3,240
Geometric Mean of NOECs			3,595	

Notes
NOEC = no-observed-effect concentration
SMAV = species mean acute value
TRV = toxicity reference value
a - USEPA (1988). Ambient ALC for nickel
b - Hunt et al. (2002)

APPENDIX C
EXPOSURE POINT CONCENTRATIONS USED FOR EXPOSURE ASSESSMENT IN THE BERA

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment	Site-wide	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	103	37	94.7	160	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	103	67	0.401	0.559	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	103	90	6.69	22.9	110
		Mercury	unk	ucl.proucl.np	mg/kg	103	94	0.117	0.258	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	103	95	6.08	7.90	17.8
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	132	100	2,390	5,660	58,300
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	132	100	776	1,840	20,400
		$TEQ_{P\ B}$ [g]	unk	ucl.proucl.np	ng/kg	18	94	4.85	13.9	28.0
		$TEQ_{P\ M}$ [h]	unk	ucl.proucl.np	ng/kg	18	94	0.902	2.23	4.50
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	18	0	3,180	13,100	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	103	100	31.8	97.6	305
	Peninsula shoreline	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	31	68	158	388	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	31	58	0.342	0.710	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	31	90	6.69	35.8	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	31	94	0.270	0.707	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	31	87	5.20	8.32	14.4
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	35	100	5,470	13,300	43,900
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	35	100	1,780	4,280	12,600
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	4	100	18.4	31.6	28.0
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	4	100	2.99	4.63	4.50
		Total PCBs (sum of Aroclors) [i]	normal	ucl.t	µg/kg	4	0	14,000	35,300	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	31	100	28.0	153	228
	Shoreline	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	44	61	119	285	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	44	66	0.342	0.608	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	44	86	5.83	25.4	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	44	93	0.197	0.512	2.02
		Nickel	unk	ucl.proucl.np	mg/kg	44	91	5.28	7.76	14.4
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	48	100	3,990	9,850	43,900
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	48	100	3,430	8,480	38,700
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	48	100	1,300	3,180	12,600
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	4	100	18.4	31.6	28.0
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	4	100	2.99	4.63	4.50
		Total PCBs (sum of Aroclors) [i]	normal	ucl.t	µg/kg	4	0	14,000	35,300	40,000
		Zinc	lognormal	ucl.cheb.log	mg/kg	44	100	25.9	111	228

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment (continued)	All Outside Western Cell	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	96	32	71.7	134	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	96	65	0.359	0.498	1.40
		Copper	lognormal	ucl.cheb.log	mg/kg	96	90	6.08	19.1	110
		Mercury	unk	ucl.proucl.np	mg/kg	96	94	0.0507	0.0918	0.717
		Nickel	unk	ucl.proucl.np	mg/kg	96	95	5.83	7.65	17.8
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	122	100	1,160	3,830	58,300
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	122	100	997	3,290	49,200
		$TEQ_{P\ B}$ [g]	unk	ucl.proucl.np	ng/kg	17	94	3.55	11.1	28.0
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	17	0	1,010	4,190	12,500
		Zinc	lognormal	ucl.cheb.log	mg/kg	96	100	29.1	84.8	305
	All Background	$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	29	100	2.52	5.62	14.7
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	11	73	0.952	1.18	1.59
		$TEQ_{P\ M}$ [k,n]	normal	ucl.t	ng/kg	8	100	0.165	0.198	0.222
	Shoreline Background	$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	8	100	1.03	3.27	4.54
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	8	100	1.08	3.30	4.55
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	100	0.400	0.607	0.952
	Post-TCRA: All Site	$TEQ_{DF\ B}$-Median [e,l]	lognormal	ucl.cheb.log	ng/kg	103	100	12.9	149	482
		$TEQ_{P\ B}$-Median [g,l]	lognormal	ucl.cheb.log	ng/kg	15	93	0.878	1.48	2.03
	Post-TCRA: Shoreline	$TEQ_{DF\ M}$-Median [f,l]	unk	ucl.proucl.np	ng/kg	33	100	2.76	5.38	14.3
		$EcoTEQ_{DF\ B}$-Median [e,j,l]	lognormal	ucl.cheb.log	ng/kg	33	100	3.08	26.5	49.6
		$TEQ_{DF\ B}$-Median [e,l]	lognormal	ucl.cheb.log	ng/kg	33	100	3.50	29.2	54.9
Soils	North of I-10	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	32	63	42.7	496	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	32	94	0.362	0.753	1.73
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	8.99	43.3	121
		Mercury	unk	ucl.proucl.np	mg/kg	32	94	0.924	3.15	12.9
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	6.07	21.7	96.0
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	42	100	1,950	6,200	30,100
		$EcoTEQ_{DF\ B}$ [e,j]	unk	ucl.proucl.np	ng/kg	42	100	1,650	5,190	24,400
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	42	100	636	2,070	11,200
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	2	100	2.16	7.05	2.93
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	2	100	0.748	2.97	1.10
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	5	20	90.0	110	108
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	46.2	257	328

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Soils (continued)	Peninsula	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	41	71	51.0	538	2,200
		Cadmium	unk	ucl.proucl.np	mg/kg	41	95	0.376	0.705	1.73
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	11.7	58.0	121
		Mercury	unk	ucl.proucl.np	mg/kg	41	95	0.733	2.48	12.9
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	6.81	22.8	96.0
		$TEQ_{DF\ B}$ [e]	unk	ucl.proucl.np	ng/kg	51	100	1,610	5,130	30,100
		$TEQ_{DF\ M}$ [f]	unk	ucl.proucl.np	ng/kg	51	100	526	1,720	11,200
		$TEQ_{P\ B}$ [g]	NA	max	ng/kg	1	100	2.93	2.93	2.93
		$TEQ_{P\ M}$ [h]	NA	max	ng/kg	1	100	1.10	1.10	1.10
		Total PCBs (sum of Aroclors) [i]	unk	ucl.proucl.np	µg/kg	24	71	47.0	79.3	119
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	67.7	516	4,160
	South of I-10, 0 to 2 feet	Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	25	88	78.9	938	2,200
		Cadmium	unk	ucl.proucl.np	mg/kg	27	100	0.336	0.591	1.28
		Chromium	lognormal	ucl.cheb.log	mg/kg	27	100	15.0	37.4	70.3
		Copper	lognormal	ucl.cheb.log	mg/kg	27	100	26.2	118	651
		Lead	lognormal	ucl.cheb.log	mg/kg	27	100	27.8	70.3	137
		Mercury	unk	ucl.proucl.np	mg/kg	27	96	0.0413	0.0765	0.156
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	30	100	6.23	31.2	105
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	30	100	5.82	16.7	38.8
		Thallium	lognormal	ucl.cheb.log	mg/kg	27	63	2.61	6.15	9.80
		Total PCBs (sum of Aroclors) [i]	lognormal	ucl.cheb.log	µg/kg	27	78	30.5	111	427
		Zinc	lognormal	ucl.cheb.log	mg/kg	27	100	178	1,260	4,160
	South of I-10, 0 to 6 inches	Barium	normal	ucl.t	mg/kg	10	100	163	226	413
		Bis(2-ethylhexyl)phthalate	lognormal	ucl.cheb.log	µg/kg	10	100	92.2	586	2,200
		Cadmium	lognormal	ucl.cheb.log	mg/kg	10	100	0.262	0.946	1.28
		Chromium	lognormal	ucl.cheb.log	mg/kg	10	100	13.3	38.1	70.3
		Copper	normal	ucl.t	mg/kg	10	100	40.5	61.2	121
		Lead	lognormal	ucl.cheb.log	mg/kg	10	100	29.1	77.9	113
		Mercury	lognormal	ucl.cheb.log	mg/kg	10	100	0.0362	0.125	0.140
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	10	100	6.78	52.3	73.1
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	10	100	11.8	18.1	31.1
		Thallium	normal	ucl.t	mg/kg	10	80	5.08	6.77	9.80
		Total PCBs (sum of Aroclors) [i]	lognormal	ucl.cheb.log	µg/kg	10	80	27.6	85.7	119
		Vanadium	normal	ucl.t	mg/kg	10	100	19.7	24.5	33.9
		Zinc	lognormal	ucl.cheb.log	mg/kg	10	100	234	1420	4,160

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Soils (continued)	Post-TCRA, North of I-10	Mercury-Median [l]	unk	ucl.proucl.np	mg/kg	24	91	1.01	3.97	12.9
		Zinc-Median [l]	lognormal	ucl.cheb.log	mg/kg	24	100	43.7	253	328
		$EcoTEQ_{DF\ B}$ [e,j,l]	lognormal	ucl.cheb.log	ng/kg	34	100	4.15	20.7	33.5
		$TEQ_{DF\ B}$ [e,l]	lognormal	ucl.cheb.log	ng/kg	34	100	4.44	22.9	34.0
	Background, North of I-10	Mercury	unk	ucl.proucl.np	mg/kg	20	100	0.0422	0.0739	0.137
		Zinc	lognormal	ucl.cheb.log	mg/kg	19	100	30.6	95.7	276
		$EcoTEQ_{DF\ B}$ [e,j,l]	unk	ucl.proucl.np	ng/kg	20	100	1.75	3.88	7.75
		$TEQ_{DF\ B}$ [e,l]	unk	ucl.proucl.np	ng/kg	20	100	1.82	3.97	7.81
Surface water	Site-wide	$TEQ_{DF\ B}$ [e,p]	NA	NA	mg/L	2	100	2.63E-08	4.00E-08	4.00E-08
Common rangia	Site-wide	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	25	0	982	1,020	1,170
		Cadmium	normal	ucl.t	mg/kg	25	100	0.246	0.257	0.303
		Copper	unk	ucl.proucl.np	mg/kg	25	100	24.6	32.6	41.4
		Mercury	normal	ucl.t	mg/kg	25	92	0.0960	0.108	0.168
		Nickel	normal	ucl.t	mg/kg	25	100	12.0	13.2	20.1
		$EcoTEQ_{DF\ B}$ [e,j]	lognormal	ucl.cheb.log	ng/kg	25	100	82.2	335	928
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	25	100	90.4	369	1,020
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	25	100	9.75	43.1	108
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	25	100	23.3	93.8	254
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	25	100	27.0	43.8	64.9
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg ww	25	100	2.91	4.52	7.40
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	25	100	3.27	5.61	15.8
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	25	100	239	356	555
		Zinc	normal	ucl.t	mg/kg	25	100	95.9	99.3	119
	Peninsula only	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	15	0	979	1,050	1,170
		Cadmium	normal	ucl.t	mg/kg	15	100	0.249	0.265	0.303
		Copper	unk	ucl.proucl.np	mg/kg	15	100	29.3	39.5	41.4
		Mercury	normal	ucl.t	mg/kg	15	100	0.105	0.122	0.168
		Nickel	unk	ucl.proucl.np	mg/kg	15	100	11.5	14.9	14.9
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	15	100	123	789	1,020
		$TEQ_{DF\ M}$ [f]	lognormal	ucl.cheb.log	ng/kg	15	100	30.7	192	254
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	15	100	29.7	55.4	64.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	15	100	3.53	4.15	5.92
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	15	100	294	463	555
		Zinc	normal	ucl.t	mg/kg	15	100	95.9	98.6	105

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia (continued)	Background	$TEQ_{DF\,B}$ [e]	normal	ucl.t	ng/kg ww	10	100	1.48	1.93	2.65
		$TEQ_{P\,B}$ [g]	normal	ucl.t	ng/kg ww	10	100	1.19	1.41	1.93
		$EcoTEQ_{DF\,B}$ [e,j]	normal	ucl.t	ng/kg	10	100	13.6	17.6	22.5
		$TEQ_{DF\,B}$ [e]	normal	ucl.t	ng/kg	10	100	14.4	18.7	23.7
		$TEQ_{DF\,M}$ [f]	normal	ucl.t	ng/kg	10	100	3.51	4.50	6.65
		$TEQ_{P\,B}$ [g]	normal	ucl.t	ng/kg	10	100	11.5	13.4	17.3
		$TEQ_{P\,M}$ [h]	normal	ucl.t	ng/kg	10	100	1.74	1.98	2.52
Gulf killifish	Site-wide	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	10	0	434	439	445
		Cadmium [n]	normal	ucl.t	mg/kg	8	13	0.00794	0.00919	0.0109
		Copper	normal	ucl.t	mg/kg	10	100	5.75	6.24	7.15
		Mercury	normal	ucl.t	mg/kg	10	100	0.202	0.263	0.372
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	1.89	2.57	3.40
		$TEQ_{DF\,B}$ [e]	lognormal	ucl.cheb.log	ng/kg	10	70	2.66	51.5	59.0
		$TEQ_{DF\,B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	10	70	0.645	12.5	14.3
		$TEQ_{DF\,M}$ [f]	lognormal	ucl.cheb.log	ng/kg	10	70	1.45	24.1	41.7
		$TEQ_{P\,B}$ [g]	lognormal	ucl.cheb.log	ng/kg	10	100	9.39	17.3	17.9
		$TEQ_{P\,B}$ [g]	normal	ucl.t	ng/kg ww	10	100	2.51	3.20	4.31
		$TEQ_{P\,M}$ [h]	lognormal	ucl.cheb.log	ng/kg	10	100	2.81	8.54	12.1
		Total PCBs [m]	unk	ucl.proucl.np	µg/kg	10	100	201	484	588
		Zinc	normal	ucl.t	mg/kg	10	100	174	180	195
	Peninsula only	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	6	0	432	441	445
		Cadmium [n,o]	normal	ucl.t	mg/kg	4	25	0.00755	0.0104	0.0109
		Copper	normal	ucl.t	mg/kg	6	100	6.04	6.63	7.15
		Mercury	normal	ucl.t	mg/kg	6	100	0.221	0.294	0.331
		Nickel	lognormal	ucl.cheb.log	mg/kg	6	100	2.06	3.16	3.40
		$TEQ_{DF\,B}$ [e]	lognormal	ucl.cheb.log	ng/kg	6	100	10.4	56.9	59.0
		$TEQ_{DF\,M}$ [f]	lognormal	ucl.cheb.log	ng/kg	6	100	4.70	38.1	41.7
		$TEQ_{P\,B}$ [g]	normal	ucl.t	ng/kg	6	100	12.9	16.9	17.9
		$TEQ_{P\,M}$ [h]	lognormal	ucl.cheb.log	ng/kg	6	100	4.09	13.6	12.1
		Total PCBs [m]	lognormal	ucl.cheb.log	µg/kg	6	100	199	686	588
		Zinc	normal	ucl.t	mg/kg	6	100	178	187	195

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Gulf killifish (continued)	Background	$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	8	88	0.258	0.401	0.636
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg ww	8	100	0.564	0.903	0.950
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	8	88	1.07	1.66	2.64
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	88	0.54	0.82	1.25
		$TEQ_{P\ B}$ [g]	lognormal	ucl.cheb.log	ng/kg	8	100	2.35	3.73	3.94
		$TEQ_{P\ M}$ [h]	lognormal	ucl.cheb.log	ng/kg	8	100	1.01	2.44	2.71
Blue Crab	Site-wide	Bis(2-ethylhexyl)phthalate [n]	unk	ucl.proucl.np	µg/kg	8	100	891	1,260	1,330
		Cadmium	normal	ucl.t	mg/kg	9	100	0.270	0.305	0.368
		Copper	normal	ucl.t	mg/kg	9	100	46.2	51.3	58.8
		Mercury	normal	ucl.t	mg/kg	9	100	0.0743	0.0854	0.0979
		Nickel	lognormal	ucl.cheb.log	mg/kg	9	100	1.24	2.79	3.66
		$EcoTEQ_{DF\ B}$ [e,j]	lognormal	ucl.cheb.log	ng/kg	9	100	19.5	38.9	40.8
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	9	100	22.2	44.7	46.3
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	9	100	7.80	10.6	14.9
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	9	100	17.9	22.3	26.8
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg ww	9	100	6.52	13.8	14.5
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	9	100	5.27	6.55	8.27
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	9	100	2.33	3.00	4.00
		Total PCBs [m]	normal	ucl.t	µg/kg	9	100	69.4	80.4	100
		Zinc	normal	ucl.t	mg/kg	9	100	112	117	123
	Background[p]	$TEQ_{DF\ B}$ [e]	NA	max	ng/kg ww	3	100	0.355	0.453	0.453
		$TEQ_{P\ B}$ [g]	NA	max	ng/kg ww	3	100	0.480	0.616	0.616
		$EcoTEQ_{DF\ B}$ [e,j]	normal	max	ng/kg	3	100	1.11	1.42	1.42
		$TEQ_{DF\ B}$ [e]	lognormal	ucl.cheb.log	ng/kg	3	100	1.18	1.55	1.55
		$TEQ_{DF\ M}$ [f]	normal	max	ng/kg	3	100	0.59	0.71	0.71
		$TEQ_{P\ B}$ [g]	normal	max	ng/kg	3	100	1.72	2.08	2.08
		$TEQ_{P\ M}$ [h]	normal	max	ng/kg	3	100	0.45	0.59	0.59

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Hardhead catfish	Site-wide	Bis(2-ethylhexyl)phthalate [n]	normal	ucl.t	µg/kg	5	80	757	1,030	1,090
		Cadmium	normal	ucl.t	mg/kg	10	80	0.0222	0.0274	0.0378
		Copper	lognormal	ucl.cheb.log	mg/kg	10	100	1.69	2.80	4.27
		Mercury	normal	ucl.t	mg/kg	10	100	0.245	0.304	0.432
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	0.937	2.46	4.36
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	10	100	79.5	93.6	117
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	10	100	27.2	32.7	44.3
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	10	100	68.7	81.6	104
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	10	100	35.4	40.3	47.4
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	10	100	12.1	14.0	17.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	10	100	22.9	27.0	30.6
		Total PCBs [m]	normal	ucl.t	µg/kg	10	100	1,480	1,680	2,010
		Zinc	normal	ucl.t	mg/kg	10	100	650	740	876
	Background	$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg ww	8	100	2.65	3.16	3.62
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg ww	8	100	4.84	5.44	6.56
		$TEQ_{DF\ B}$ [e]	normal	ucl.t	ng/kg	8	100	7.73	9.13	10.6
		$TEQ_{DF\ M}$ [f]	normal	ucl.t	ng/kg	8	100	6.51	7.58	8.11
		$TEQ_{P\ B}$ [g]	normal	ucl.t	ng/kg	8	100	14.2	15.8	18.9
		$TEQ_{P\ M}$ [h]	normal	ucl.t	ng/kg	8	100	7.80	9.54	12.4
Terrestrial invertebrates[r]	North of I-10	Cadmium	unk	ucl.proucl.np	mg/kg	32	94	3.69	6.61	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	4.63	22.3	NA
		Mercury [s]	unk	ucl.proucl.np	mg/kg	32	94	0.960	2.62	9.03
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	0.759	2.71	NA
		Total PCBs [i]	unk	ucl.proucl.np	mg/kg	5	20	0.155	0.198	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	301	528	NA
		$EcoTEQ_{DF\ B}$ [j,q]	unk	ucl.proucl.np	ng/kg	42	100	60.7	181	900
		$TEQ_{DF\ B}$ [q]	unk	ucl.proucl.np	ng/kg	42	100	117	359	1,840
	Peninsula only	Cadmium	unk	ucl.proucl.np	mg/kg	41	95	3.81	6.27	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	6.03	29.9	NA
		Mercury [r]	unk	ucl.proucl.np	mg/kg	41	95	0.788	2.10	9.03
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	0.851	2.85	NA
		Total PCBs [i]	unk	ucl.proucl.np	mg/kg	24	71	0.0639	0.130	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	341	664	NA
		$TEQ_{DF\ M}$ [q]	unk	ucl.proucl.np	ng/kg	51	100	94.2	284	1770

Table C-1
Exposure Point Concentrations Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units [a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Terrestrial plants[t]	North of I-10	Cadmium	unk	ucl.proucl.np	mg/kg	32	94	0.357	0.533	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	32	100	1.22	2.26	NA
		Mercury	unk	ucl.proucl.np	mg/kg	32	94	0.0347	0.118	NA
		Nickel	lognormal	ucl.cheb.log	mg/kg	32	97	0.417	1.08	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	32	100	40.4	105	NA
	Peninsula	Cadmium	unk	ucl.proucl.np	mg/kg	41	95	0.365	0.514	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	41	100	1.35	2.54	NA
		Mercury	unk	ucl.proucl.np	mg/kg	41	95	0.0275	0.0930	NA
		Nickel	lognormal	ucl.cheb.log	mg/kg	41	98	0.455	1.12	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	41	100	49.9	154	NA
Aquatic plants[t]	Shoreline	Cadmium	unk	ucl.proucl.np	mg/kg	44	66	0.346	0.474	NA
		Copper	lognormal	ucl.cheb.log	mg/kg	44	86	1.03	1.83	NA
		Mercury	unk	ucl.proucl.np	mg/kg	44	93	0.00740	0.0192	NA
		Nickel	unk	ucl.proucl.np	mg/kg	44	91	0.376	0.502	NA
		Zinc	lognormal	ucl.cheb.log	mg/kg	44	100	29.3	65.7	NA

Notes

BERA - Baseline Ecological Risk Assessment
CT = central tendency
EPC = exposure point concentration
NA = not applicable
RM = reasonable maximum
ROS = regression on order statistics, a method for substituting for non-detects
TCRA = time critical removal action
UCL = upper confidence limit on the mean
ucl.t = UCL for normally distributed data, calculated based on the T statistic
ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor
ucl.proucl.np = nonparametric UCL for an unknown data distribution, method is based on that used in ProUCL (USEPA 2010)
unk = unknown distribution

a - All concentrations are on a dry weight basis unless the units indicate otherwise.
b - The mean value is the CT EPC.
c - The UCL will be used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration will be selected as the RM EPC.
d - Because the detection frequency was between 20 and 50% and N > 10, ROS was used for calculating the UCL.
e - Toxicity equivalent for dioxins and furans calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.
f - Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
g - Toxicity equivalent for polychlorinated biphenyls calculated using avian toxicity equivalency factors with nondetects set at one-half the detection limit.
h - Toxicity equivalent for polychlorinated biphenyls calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
i - Sum of total Aroclors with nondetects set at one-half the detection limit.
j - Calculated using a relative bioavailability adjustment factor for avian receptors for 2,3,7,8-TCDD.
k - Toxicity equivalent for polychlorinated biphenyls calculated using mammalian toxicity equivalency factors with nondetects set at one-half the detection limit.
l - Median of background data (soils or sediments, as appropriate) used to substitute for samples in the TCRA footprint.
m - Sum of 43 PCB congeners with non-detects set at one-half the detection limit.
n - High-biasing nondetects (nondetects > highest detected value) were removed prior to calculating the EPC.
o - Detection frequency between 20 and 50% but N<10, so no ROS performed for this dataset.
p - Data set too small to generate a UCL; maximum value is used for the RM.
q - Estimated using site soil data and regression relationships (see Appendix D).
r - Unless otherwise footnoted, estimated using site data for soils from indicated exposure area and soil-to-invertebrate BAFs
s - Estimated using site soil data and variable BAFs depending on concentration in soils (Burton et al. 2006)
t - Estimated using site data for soils or sediments from indicated exposure area and soil-to-plant BAFs

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment	FCA1	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	29	52	33.2	55.8	120
		Cadmium	lognormal	ucl.cheb.log	mg/kg	29	79	0.345	0.815	1.40
		Copper	lognormal	ucl.cheb.log	mg/kg	29	90	7.51	31.9	110
		Mercury	lognormal	ucl.cheb.log	mg/kg	29	93	0.0277	0.0992	0.0960
		Nickel	lognormal	ucl.cheb.log	mg/kg	29	97	5.16	12.7	15.7
		Zinc	lognormal	ucl.cheb.log	mg/kg	29	100	39.5	132	305
	FCA2	Bis(2-ethylhexyl)phthalate [d]	unk	ucl.proucl.np	µg/kg	57	37	103	162	1,600
		Cadmium	unk	ucl.proucl.np	mg/kg	57	63	0.374	0.602	1.60
		Copper	lognormal	ucl.cheb.log	mg/kg	57	93	7.18	21.4	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	57	95	0.179	0.428	2.02
		Nickel	lognormal	ucl.cheb.log	mg/kg	57	100	4.61	9.64	17.8
		Zinc	lognormal	ucl.cheb.log	mg/kg	57	100	32.1	88.5	228
	FCA3	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	17	12	192	957	3,000
		Cadmium	unk	ucl.proucl.np	mg/kg	17	59	0.394	0.725	1.00
		Copper	lognormal	ucl.cheb.log	mg/kg	17	82	4.34	33.3	19.0
		Mercury	unk	ucl.proucl.np	mg/kg	17	94	0.0420	0.0792	0.0900
		Nickel	unk	ucl.proucl.np	mg/kg	17	76	5.33	10.1	11.5
		Zinc	lognormal	ucl.cheb.log	mg/kg	17	100	21.3	167	97.2
Common rangia	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	1,050	1,110	1,130
		Cadmium	normal	ucl.t	mg/kg	5	100	0.257	0.283	0.290
		Copper	normal	ucl.t	mg/kg	5	100	17.4	18.3	18.8
		Mercury	normal	ucl.t	mg/kg	5	100	0.0946	0.117	0.127
		Nickel	normal	ucl.t	mg/kg	5	100	16.5	19.1	20.1
		Zinc	normal	ucl.t	mg/kg	5	100	107	116	119
	FCA2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	15	0	913	960	1,100
		Cadmium	normal	ucl.t	mg/kg	15	100	0.237	0.252	0.303
		Copper	lognormal	ucl.cheb.log	mg/kg	15	100	22.8	35.6	41.4
		Mercury	normal	ucl.t	mg/kg	15	87	0.0837	0.100	0.136
		Nickel	normal	ucl.t	mg/kg	15	100	10.4	11.7	14.2
		Zinc	normal	ucl.t	mg/kg	15	100	93.1	97.2	105

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia (continued)	FCA3	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	5	0	1,120	1,260	1,170
		Cadmium	normal	ucl.t	mg/kg	5	100	0.262	0.286	0.290
		Copper	normal	ucl.t	mg/kg	5	100	32.0	32.5	32.7
		Mercury	normal	ucl.t	mg/kg	5	100	0.134	0.154	0.168
		Nickel	normal	ucl.t	mg/kg	5	100	12.3	14.9	14.9
		Zinc	normal	ucl.t	mg/kg	5	100	92.8	98.6	101
Gulf killifish	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	443
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00940	N/A	0.00970
		Copper	normal	ucl.t	mg/kg	2	100	5.43	N/A	5.58
		Mercury	normal	ucl.t	mg/kg	2	100	0.117	N/A	0.136
		Nickel	normal	ucl.t	mg/kg	2	100	1.58	N/A	1.63
		Zinc	normal	ucl.t	mg/kg	2	100	168	N/A	176
	FCA2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	6	0	431	440	445
		Cadmium [e,f]	normal	ucl.t	mg/kg	4	25	0.00789	0.0104	0.0109
		Copper	normal	ucl.t	mg/kg	6	100	5.71	6.60	7.15
		Mercury	normal	ucl.t	mg/kg	6	100	0.206	0.302	0.372
		Nickel	normal	ucl.t	mg/kg	6	100	1.80	1.94	2.03
		Zinc	normal	ucl.t	mg/kg	6	100	171	174	176
	FCA3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	439
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00657	N/A	0.00771
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	6.40
		Mercury	normal	ucl.t	mg/kg	2	100	0.278	N/A	0.319
		Nickel	normal	ucl.t	mg/kg	2	100	2.73	N/A	3.40
		Zinc	normal	ucl.t	mg/kg	2	100	191	N/A	195
Blue crab	FCA1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	3	100	814	N/A	894
		Cadmium	normal	ucl.t	mg/kg	3	100	0.273	N/A	0.300
		Copper	normal	ucl.t	mg/kg	3	100	45.3	N/A	55.8
		Mercury	normal	ucl.t	mg/kg	3	100	0.0888	N/A	0.0979
		Nickel	normal	ucl.t	mg/kg	3	100	2.51	N/A	3.66
		Zinc	normal	ucl.t	mg/kg	3	100	117	N/A	122

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Blue crab (continued)	FCA2	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	2	100	970	N/A	1,170
		Cadmium	normal	ucl.t	mg/kg	3	100	0.318	N/A	0.368
		Copper	normal	ucl.t	mg/kg	3	100	48.1	N/A	58.8
		Mercury	normal	ucl.t	mg/kg	3	100	0.0568	N/A	0.0693
		Nickel	normal	ucl.t	mg/kg	3	100	0.829	N/A	1.03
		Zinc	normal	ucl.t	mg/kg	3	100	113	N/A	123
	FCA3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	3	100	914	N/A	1,330
		Cadmium	normal	ucl.t	mg/kg	3	100	0.220	N/A	0.274
		Copper	normal	ucl.t	mg/kg	3	100	45.3	N/A	48.9
		Mercury	normal	ucl.t	mg/kg	3	100	0.0772	N/A	0.0974
		Nickel	normal	ucl.t	mg/kg	3	100	1.13	N/A	1.74
		Zinc	normal	ucl.t	mg/kg	3	100	106	N/A	112
Hardhead catfish	FCA1	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	2	100	999	N/A	1,090
		Cadmium	normal	ucl.t	mg/kg	3	100	0.0264	N/A	0.0378
		Copper	normal	ucl.t	mg/kg	3	100	1.72	N/A	2.31
		Mercury	normal	ucl.t	mg/kg	3	100	0.300	N/A	0.338
		Nickel	normal	ucl.t	mg/kg	3	100	0.786	N/A	1.22
		Zinc	normal	ucl.t	mg/kg	3	100	762	N/A	876
	FCA2	Bis(2-ethylhexyl)phthalate [e]	normal	ucl.t	µg/kg	3	67	595	N/A	833
		Cadmium	normal	ucl.t	mg/kg	4	75	0.0206	0.0300	0.0289
		Copper	normal	ucl.t	mg/kg	4	100	1.39	1.72	1.78
		Mercury	normal	ucl.t	mg/kg	4	100	0.252	0.405	0.432
		Nickel	normal	ucl.t	mg/kg	4	100	0.846	1.43	1.49
		Zinc	normal	ucl.t	mg/kg	4	100	528	711	748

Table C-2
Exposure Point Concentrations for Individual FCAs Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution Type	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Hardhead catfish (continued)	FCA3	Bis(2-ethylhexyl)phthalate [e,g]	normal	ucl.t	µg/kg	3	0	331	N/A	331
		Cadmium	normal	ucl.t	mg/kg	3	67	0.0202	N/A	0.0323
		Copper	normal	ucl.t	mg/kg	3	100	2.53	N/A	4.27
		Mercury	normal	ucl.t	mg/kg	3	100	0.181	N/A	0.254
		Nickel	normal	ucl.t	mg/kg	3	100	2.21	N/A	4.36
		Zinc	normal	ucl.t	mg/kg	3	100	701	N/A	782

Notes
CT = central tendency
EPC = exposure point concentration
FCA = fish collection area
N/A = not applicable
RM = reasonable maximum
ROS = regression on order statistics, a method for substituting for non-detects
UCL = upper confidence limit on the mean
ucl.t = UCL for normally distributed data, calculated based on the T statistic
ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor
ucl.proucl.np = nonparametric UCL for an unknown data distribution, same method as ProUCL (USEPA 2010)
unk = unknown distribution
a - All concentrations are on a dry weight basis unless the units indicate otherwise.
b - The mean value is the CT EPC.
c - The UCL will be used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration will be selected as the RM EPC. For N ≤ 3, a UCL cannot be calculated, so the average of the two samples and the maximum value only are reported in these cases.
d - Because the detection frequency was between 20 and 50% and N >10, ROS was used for calculating the UCL.
e - High-biasing nondetects (nondetects > highest detected value) were removed prior to calculating the EPC.
f - Detection frequency between 20 and 50% but N<10, so no ROS performed for this data set.
g - The highest detection limit for all FCAs was used to generate statistics.

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Sediment	TTR1 and TTR2 [d]	Bis(2-ethylhexyl)phthalate [e]	lognormal	ucl.cheb.log	µg/kg	7	43	22.5	84.4	73.0
		Cadmium	normal	ucl.t	mg/kg	7	100	0.431	0.542	0.700
		Copper	normal	ucl.t	mg/kg	7	100	7.82	10.1	11.5
		Mercury	normal	ucl.t	mg/kg	7	100	0.0330	0.0445	0.0560
		Nickel	normal	ucl.t	mg/kg	7	100	7.60	9.49	12.0
		Zinc	normal	ucl.t	mg/kg	7	100	39.4	54.4	65.0
	TTR3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	10	90	180	249	350
		Cadmium	unk	ucl.proucl.np	mg/kg	10	60	0.346	1.01	1.50
		Copper	lognormal	ucl.cheb.log	mg/kg	10	100	10.6	30.1	65.6
		Mercury	unk	ucl.proucl.np	mg/kg	10	100	0.295	1.16	2.02
		Nickel	lognormal	ucl.cheb.log	mg/kg	10	100	4.26	9.42	14.1
		Zinc	lognormal	ucl.cheb.log	mg/kg	10	100	38.2	113	197
	TTR4	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	6	0	11.4	18.9	20.0
		Cadmium [e]	unk	ucl.proucl.np	mg/kg	6	33	0.200	0.556	0.600
		Copper	lognormal	ucl.cheb.log	mg/kg	6	50	1.73	8.94	15.2
		Mercury	unk	ucl.proucl.np	mg/kg	6	100	0.0153	0.0493	0.0530
		Nickel	lognormal	ucl.cheb.log	mg/kg	6	100	1.80	6.48	10.0
		Zinc	lognormal	ucl.cheb.log	mg/kg	6	100	11.6	51.6	74.4
	TTR5	Bis(2-ethylhexyl)phthalate [f]	lognormal	ucl.cheb.log	µg/kg	12	33	19	37.1	76.0
		Cadmium	unk	ucl.proucl.np	mg/kg	12	50	0.138	0.383	0.700
		Copper	normal	ucl.t	mg/kg	12	100	6.08	7.53	11.8
		Mercury	lognormal	ucl.cheb.log	mg/kg	12	83	0.0130	0.0330	0.0520
		Nickel	lognormal	ucl.cheb.log	mg/kg	12	100	5.18	8.61	12.5
		Zinc	lognormal	ucl.cheb.log	mg/kg	12	100	20.9	48.3	55.4
	TTR6	Bis(2-ethylhexyl)phthalate	all below DL	max	µg/kg	5	0	9.50	9.50	9.50
		Cadmium	all below DL	max	mg/kg	5	0	0.100	0.100	0.100
		Copper [e]	lognormal	ucl.cheb.log	mg/kg	5	40	0.812	4.26	3.50
		Mercury	normal	ucl.t	mg/kg	5	100	0.00590	0.0104	0.0140
		Nickel	normal	ucl.t	mg/kg	5	20	0.315	0.377	0.425
		Zinc	lognormal	ucl.cheb.log	mg/kg	5	100	3.35	8.61	9.00

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Common rangia	TTR1	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	1,050	1,110	1,130
		Cadmium	normal	ucl.t	mg/kg	5	100	0.257	0.283	0.290
		Copper	normal	ucl.t	mg/kg	5	100	17.4	18.3	18.8
		Mercury	normal	ucl.t	mg/kg	5	100	0.0946	0.117	0.127
		Nickel	normal	ucl.t	mg/kg	5	100	16.5	19.1	20.1
		Zinc	normal	ucl.t	mg/kg	5	100	107	116	119
	TTR3	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	954	1,050	1,100
		Cadmium	lognormal	ucl.cheb.log	mg/kg	5	100	0.274	0.321	0.303
		Copper	normal	ucl.t	mg/kg	5	100	38.1	41.2	41.4
		Mercury	normal	ucl.t	mg/kg	5	100	0.124	0.135	0.136
		Nickel	unk	ucl.proucl.np	mg/kg	5	100	12.5	16.8	14.2
		Zinc	normal	ucl.t	mg/kg	5	100	98.8	104	105
	TTR4	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	920	985	995
		Cadmium	normal	ucl.t	mg/kg	5	100	0.226	0.243	0.252
		Copper	normal	ucl.t	mg/kg	5	100	17.6	19.8	19.8
		Mercury	normal	ucl.t	mg/kg	5	60	0.0700	0.101	0.108
		Nickel	lognormal	ucl.cheb.log	mg/kg	5	100	8.90	11.8	11.7
		Zinc	normal	ucl.t	mg/kg	5	100	84.6	92.5	97.6
	TTR5	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	5	0	865	995	1,000
		Cadmium	normal	ucl.t	mg/kg	5	100	0.210	0.227	0.227
		Copper	lognormal	ucl.cheb.log	mg/kg	5	100	17.7	21.3	21.1
		Mercury	normal	ucl.t	mg/kg	5	100	0.0568	0.0666	0.0657
		Nickel	normal	ucl.t	mg/kg	5	100	9.57	13.0	13.3
	TTR6	Bis(2-ethylhexyl)phthalate	unk	ucl.proucl.np	µg/kg	5	0	1120	1260	1170
		Cadmium	normal	ucl.t	mg/kg	5	100	0.262	0.286	0.290
		Copper	normal	ucl.t	mg/kg	5	100	32.0	32.5	32.7
		Mercury	normal	ucl.t	mg/kg	5	100	0.134	0.154	0.168
		Nickel	normal	ucl.t	mg/kg	5	100	12.3	14.9	14.9
		Zinc	normal	ucl.t	mg/kg	5	100	92.8	98.6	101

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
Gulf killifish	TTR2	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	443
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00940	N/A	0.00970
		Copper	normal	ucl.t	mg/kg	2	100	5.43	N/A	5.58
		Mercury	normal	ucl.t	mg/kg	2	100	0.117	N/A	0.136
		Nickel	normal	ucl.t	mg/kg	2	100	1.58	N/A	1.63
		Zinc	normal	ucl.t	mg/kg	2	100	168	N/A	176
	TTR3 [g]	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	439	N/A	445
		Copper	normal	ucl.t	mg/kg	2	100	5.75	N/A	6.16
		Mercury	normal	ucl.t	mg/kg	2	100	0.251	N/A	0.331
		Nickel	normal	ucl.t	mg/kg	2	100	1.99	N/A	2.03
		Zinc	normal	ucl.t	mg/kg	2	100	170	N/A	175
	TTR4	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	435	N/A	436
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00724	N/A	0.00806
		Copper	normal	ucl.t	mg/kg	2	100	5.20	N/A	6.36
		Mercury	normal	ucl.t	mg/kg	2	100	0.232	N/A	0.372
		Nickel	normal	ucl.t	mg/kg	2	100	1.73	N/A	1.74
		Zinc	normal	ucl.t	mg/kg	2	100	170	N/A	171
	TTR5	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	418	N/A	420
		Cadmium	normal	ucl.t	mg/kg	2	50	0.00853	N/A	0.0109
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	7.15
		Mercury	normal	ucl.t	mg/kg	2	100	0.135	N/A	0.145
		Nickel	normal	ucl.t	mg/kg	2	100	1.68	N/A	1.81
		Zinc	normal	ucl.t	mg/kg	2	100	171	N/A	176

Table C-3
Exposure Point Concentrations for Data from Individual Sample Collection Transects Used for Exposure Assessment in the BERA

Medium	Exposure Area	Analyte	Distribution	Method	Units[a]	Number of Samples	Detection Frequency (percent)	Mean[b]	UCL[c]	Maximum Concentration
	TTR6	Bis(2-ethylhexyl)phthalate	normal	ucl.t	µg/kg	2	0	438	N/A	439
		Cadmium	normal	ucl.t	mg/kg	2	0	0.00657	N/A	0.00771
		Copper	normal	ucl.t	mg/kg	2	100	6.18	N/A	6.40
		Mercury	normal	ucl.t	mg/kg	2	100	0.278	N/A	0.319
		Nickel	normal	ucl.t	mg/kg	2	100	2.73	N/A	3.40
		Zinc	normal	ucl.t	mg/kg	2	100	191	N/A	195

Notes

CT = central tendency
EPC = exposure point concentration
N/A = not applicable
RM = reasonable maximum
ROS = regression on order statistics, a method for substituting for non-detects
TTR = transect
UCL = upper confidence limit on the mean
ucl.t = UCL for normally distributed data, calculated based on the T statistic
ucl.cheb.log = UCL for lognormally distributed data, using a Chebyshev correction factor
ucl.proucl.np = nonparametric UCL for an unknown data distribution, same method as ProUCL (USEPA 2010)
unk= unknown distribution

a - All concentrations are on a dry weight basis unless the units indicate otherwise.
b - The mean value will be used as the CT EPC.
c - The UCL is used as the RM EPC, except where UCL>maximum concentration, in which case the maximum concentration is selected as the RM EPC. For N ≤ 3, a UCL cannot be calculated, so the average of the two samples and the maximum value only are reported in these cases.
d - Transects 1 and 2 had too much overlap to create distinct data sets.
e - Detection frequency between 20 and 50% but N<10, so no ROS performed for this data set
f - Because the detection frequency was between 20 and 50% and N >10, ROS was used for calculating the UCL.
g - No cadmium data are available for Gulf killifish in transect 3 due to removal of high-biasing nondetects. Cadmium killifish data from TTR5 were used in cadmium exposure assessment for TTR3.

APPENDIX D
ESTIMATION OF DIOXIN AND FURAN CONCENTRATIONS IN TERRESTRIAL INVERTEBRATE TISSUE FOR THE EXPOSURE MODEL

ESTIMATION OF DIOXIN AND FURAN CONCENTRATIONS IN TERRESTRIAL INVERTEBRATE TISSUE FOR THE EXPOSURE MODEL

SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

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International Paper Company

U.S. Environmental Protection Agency, Region 6

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
$COPC_E$	chemical of potential ecological concern
CV	coefficient of variation
EPC	exposure point concentration
Site	San Jacinto River Waste Pits Superfund site
TEF	toxicity equvalency factor
TEQ	toxicity equivalent
TEQ_{DF}	toxicity equivalent for dioxins and furans
$TEQ_{DF,B}$	toxicity equivalent for dioxins and furans using avian TEFs
$TEQ_{DF,M}$	toxicity equivalent for dioxins and furans using mammalian TEFs
USEPA	U.S. Environmental Protection Agency

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1 INTRODUCTION

This Appendix describes the methods used to generate toxicity equivalent (TEQ) concentrations for dioxins and furans (TEQ_{DF}) in terrestrial invertebrate tissues for use in the baseline ecological risk assessment (BERA) exposure model. No empirical data are available for dioxins and furans in terrestrial invertebrate tissue at the Site, so a modeled approach is needed.

Unlike the approach used for other chemicals of potential ecological concern ($COPC_{E}$s) at the Site, a single regression equation cannot be used to estimate TEQ_{DF} because the individual dioxin congeners cannot be assumed to have the same rates or patterns of bioaccumulation (Matscheko 2002; MACTEC 2004; Integral 2010). A review of the scientific literature and a search of ecological risk assessment documents was conducted to identify sources of colocated soil and earthworm tissue dioxin and furan data that could be used for developing congener-specific uptake relationships. Although there are some published uptake factors (derived as the ratio of concentrations in earthworms to those in soil as for a BAF) for a few common dioxin congeners in the literature, notably 2,3,7,8-TCDD (Sample et al. 1998), available uptake factors cannot be extrapolated across all dioxin and furan congeners to estimate invertebrate tissue concentrations for all of the individual congeners. This was demonstrated in the Tittabawassee risk assessment when use of simplified uptake factors from the literature of 5 for 2,3,7,8-TCDD and 0.1 for 2,3,7,8-TCDF led to a 10-fold overprediction of dioxin and furan concentrations in soil invertebrates relative to measured values (Galbraith and MDEQ 2004, Kay et al. 2005). Other CERCLA sites have used data sets of similar sizes to evaluate soil-earthworm relationships and have shown that uptake factors are variable across congeners. For example, the Centredale ecological risk assessment (MACTEC 2004) used a data set of N=11 with detection frequencies between about 30% and 100% for individual congeners to describe a range from < 0.01 to 0.7 for soil-to-earthworm uptake factors for individual dioxin congeners (MACTEC 2004). Unfortunately, earthworms were not depurated in this study and therefore the data could not be used in this BERA to establish an estimate for uptake into tissues.

Data from a published U.S. Environmental Protection Agency-accepted study of dioxins in earthworm tissue and colocated soils from the Cass Lake Superfund site (Integral 2007) were used to generate significant regression relationships for individual dioxin and furan congeners. These regressions were applied to this BERA using Site soils data to estimate concentrations of congeners in terrestrial invertebrate tissue. Toxicity equivalency factors (TEFs) (Van den Berg et al. 2006) were then used to calculate TEQs for individual dioxin and furan congeners that were summed to obtain TEQ_{DF}s for terrestrial invertebrates at the Site. Exposure point concentrations (EPCs) for the resulting TEQ_{DF}s were calculated, and used in the wildlife exposure models for surrogate receptors at the Site that eat terrestrial invertebrates (killdeer and raccoon).

The remainder of this Appendix describes the details of the methods used to develop statistical relationships used to estimate the TEQ_{DF} in terrestrial invertebrate tissue, and the derivation of the tissue estimates. This includes a) the derivation of relationships between soil and earthworm tissue for individual congeners; b) the estimation of dioxin and furan concentrations in tissue for those congeners without a statistically significant relationship to soil concentration; and c) the application of these relationships to calculate a TEQ_{DF} for relevant taxa and exposure areas used in the exposure model (Section 3).

2 DERIVATION OF EQUATIONS DESCRIBING ESTIMATES OF INDIVIDUAL CONGENER CONCENTRATIONS IN TERRESTRIAL INVERTEBRATES

This section describes 1) the data used to develop soil-invertebrate tissue relationships; 2) the methods used to derive regression equations for individual congeners; and 3) the methods used to select an alternative approach for estimating a congener's concentration in tissue when no statistically significant soil-to-tissue regression equation could be identified.

2.1 Data Used for Developing Soil-Invertebrate Tissue Relationships

Available literature and Superfund-related reports evaluating dioxin and furan uptake into terrestrial invertebrates, either did not provide congener-specific data or did not use methods that would allow for development of tissue-specific uptake estimates (Section 1). The data used to develop soil-invertebrate tissue relationships for this risk assessment are from a study evaluating dioxins and furans in earthworms and soils at the St. Regis Paper Company Superfund Site in Cass Lake, Minnesota (Cass Lake site) (Integral 2007). The data set consisted of four co-located soil and earthworm tissue samples[1] and two additional soil-earthworm paired samples from a 28-day laboratory bioacummulation study. All soil samples were collected from the top 12 inches below ground surface. Undecomposed plant materials were removed from the surface prior to collection of soil samples, and a homogenized composite of five subsamples from a single location was prepared for each sampling location. Earthworm samples collected from each soil sampling location were composited into a sample of at least 50 g and were depurated in the laboratory for 24 hours to eliminate soils and other gut contents from the worms prior to analysis. Two additional soil locations were selected for the laboratory bioaccumulation study because these soils appeared suitable for invertebrates but insufficient earthworm sample mass was available at these two stations. Therefore, a 28-day earthworm bioaccumulation study using *Eisenia fetida* was used (ASTM Method E 1676-84) and dioxins and furans were measured in soils and the tested earthworm samples from these locations. These two sets of data (Table D-1) yielded six co-located samples that were used to develop regressions. All data were validated and reported for this study according to standard protocols for Superfund sites (Integral 2007).

Soil data were reported on a mg/kg dry weight basis; because the exposure model requires concentrations in receptor prey in dry weight units, earthworm data originally reported on a mg/kg wet weight basis were converted to dry weight using the following equation:

$$C_{E,dw} = C_{E,ww} \div (f_{solids})$$

Where:

$C_{E,dw}$	=	concentration in the tissue of the earthworm, dry weight (mg/kg)
$C_{E,ww}$	=	concentration in the tissue of the earthworm, wet weight (mg/kg)
f_{solids}	=	fraction of the organism that is solid material (not water).

Solids data were available for each of the samples except ECO-07. For the ECO-07 sample, the overall average of f_{solids} of the other samples was used as an estimate for this sample.

The range of most dioxin and furan concentrations in soil at the Cass Lake site were similar to the range of concentrations in soils from the San Jacinto River Waste Pits site

[1] Five colocated soil and earthworm samples (field-collected) were available from the study. However, earthworms in one sample were not depurated and had substantively higher concentrations of all congeners than other earthworms, likely attributable to soil remaining in the gut contents. This sample was therefore not included in this analysis.

(Figure D-1). In particular, dioxin and furan congener distributions in soils of Cass Lake are similar to or higher than the ranges of concentrations of congeners in those San Jacinto site soils collected from locations outside of the 1966 perimeter of the waste impoundments north of I-10 (Figure D-2). Therefore, for the majority of congeners, predictions made on the basis of Cass Lake soil concentration data are not outside of the range of San Jacinto site soils,supporting the premise that the Cass Lake dataset is appropriate for use in generating regression relationships that can be applied to the Site data. Two exceptions are 2,3,7,8-TCDD and 2,3,7,8-TCDF, which have higher concentrations in SJRWP soils within the impoundments relative to Cass Lake soil concentrations (Figure D-3). Different approaches to treatment of these congeners in modeling from soils inside and outside of the 1966 impoundment perimeter are discussed further below.

2.2 Derivation of Regressions for Estimating Concentrations of Dioxin and Furan Congeners in Invertebrate Tissue

Concentrations of individual dioxin and furan congeners in earthworm tissue from the Cass Lake datasets were regressed against soil concentration data. The distribution of the dataset was evaluated for each congener in soil and earthworm tissue, and nearly all soil and earthworm congener datasets were found to have a lognormal distribution. Both untransformed and log-transformed regressions were evaluated and in all cases, log–log relationships had similar or lower p values and similar R^2 values, supporting the assumption that log-transformed relationships are the best models for these datasets. P-values ≤ 0.1 were considered statistically significant; this p value is used because of the small size of the sample set and consequently lower power (Sokal and Rohlf 1981; Royall 1986). Significant regression relationships between soil and tissue could be developed for 11 of the 17 congeners (Table D-2). The approach taken to estimate tissue concentration for the remaining six congeners is described in the next section.

2,3,7,8-TCDD was not detected in five of six Cass Lake earthworm tissue samples. In Cass Lake soil, 2,3,7,8-TCDD was not detected in three samples. There is consequently some uncertainty regarding the soiltissue relationship for TCDD given the censored data; however, a significant regression relationship was derived for this congener (Figure D-4) when half of the detection limits were used for the undetected values. The fact that TCDD in soil was detected with colocated tissue samples in which it was not detected suggests that the uptake rate from soil to earthworm tissues is low for this congener. Because of the uncertainties of the censored data for TCDD, this regression for TCDD was not used in evaluation of correlates for congeners without regression relationships (Section 2.3 below).

2.3 Development of Estimates Using Correlated Congeners for Cases when Significant Regressions Could Not Be Identified

For six congeners, no statistically significant relationships between soil and earthworm concentrations were identified (Table D-2). Therefore, an alternative approach was used to estimate concentrations of these congeners in earthworm tissue. Spearman correlations[2] were used to evaluate relationships of each of these congeners in earthworm tissue with each of the other 11 congeners in earthworm tissue. Those congeners that had the highest (Table D-3), statistically significant (Table D-4) correlation coefficients with each of the six congeners were further evaluated in order to select the congener that provided the best correlate for each of the six congeners of interest.

To identify the best correlate, ratios were first calculated for the colocated pairs of earthworm tissue data for the congener of interest and each of its potentially well-correlated congeners. The average ratio and the coefficient of variation (CV) was then calculated for each congener pair (Table D-5). Because more than one congener was identified as a potential correlate in all cases, the congener pair with the lowest CV was selected as the best fit (Table D-5). The concentration of the congener of interest was then estimated by taking the concentration of its selected correlate and multiplying it by the mean ratio.

As discussed above, the range of 2,3,7,8-TCDF concentrations in Cass Lake soil is similar to the range of concentrations in San Jacinto site soils outside of the waste impoundments north of I-10, but the highest Cass Lake concentrations of TCDF are lower than those from within the northern impoundments. 2,3,7,8-TCDF in Cass Lake soils was found to be significantly and strongly correlated with 1,2,3,6,7,8-HxCDD in Cass Lake tissues (Table D-3). In addition, concentrations of 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD in soils outside the impoundments are similar. This similarity indicates that use of 1,2,3,6,7,8-HxCDD as a correlate would not

[2] Spearman's non-parametric ranked correlations (rho), for evaluating statistical dependence between variables.

lead to underpredictions of 2,3,7,8-TCDF in tissue (Figure D-2). However, concentrations of 1,2,3,6,7,8-HxCDD were low relative to 2,3,7,8-TCDF in soils inside the waste impoundments (Figure D-3), so a different congener was needed for prediction of TCDF concentrations in tissue from concentrations in soils inside the impoundments. 1,2,3,4,7,8-HxCDF, also significantly correlated with 2,3,7,8-TCDF (Table D-3), has concentrations that were more consistent with 2,3,7,8-TCDF within the waste impoundment soils (Figure D-3). Therefore, this congener was applied as the selected correlate for 2,3,7,8-TCDF for soil samples within the waste impoundments.

3 ESTIMATION OF TEQS IN TERRESTRIAL INVERTEBRATE TISSUE AT THE SITE

Surface soil data for the San Jacinto Site were used with the regressions and correlated congener ratios described in Section 2 to estimate $TEQ_{DF,B}$ and $TEQ_{DF,M}$ in terrestrial invertebrates at the Site. This section describes 1) how the Site soil data was selected, and 2) how the TEQ calculations were performed to generate these estimates.

3.1 Generation of Site Soil Data for Use in TEQ Calculations

Calculation of an estimated TEQ concentration in terrestrial invertebrates requires that Site soil data be used as the input variable to the individual congener regressions. Site soil datasets selected for calculation of TEQs were surface soil samples within the exposure units identified in the BERA for upland receptors whose diets include terrestrial invertebrate prey: raccoon (Figure 4-10 in the BERA) and killdeer (Figure 4-9 in the BERA).

3.2 Calculation of TEQs and Terrestrial Invertebrate EPCs

The congener-specific regression equations (Section 2.2) and correlations (Section 2.3) were used with corresponding congener concentrations in individual soil samples (Section 3.1) to estimate individual congener concentrations in the tissue of soil invertebrates.[3] For each soil sample, the result is an estimate of the concentration of each congener in a hypothetical corresponding invertebrate sample. Resulting congener-specific concentrations in modeled earthworm tissue were then multiplied by the avian or mammalian TEF, as appropriate, to

[3] For 2,3,7,8-TCDD, an additional step was taken to adjust for known limitations in bioavailability of this congener to avian receptors, using a relative bioavailability adjustment factor. Results are presented both with and without a bioavailability adjustment factor in the uncertainty analysis. See the main text, Section 4, for details.

compute the TEQ_{DF} concentration for a modeled individual earthworm sample. The final result was a set of estimated $TEQ_{DF,B}$ and $TEQ_{DF,M}$ concentrations for modeled earthworm samples, each corresponding to a specific soil sample. From the sets of estimated earthworm $TEQ_{DF,B}$ and $TEQ_{DF,M}$ concentrations, central tendency and reasonable maximum exposure point concentrations were calculated using the approach described in Section 3.8.2 of the BERA to generate estimates of terrestrial invetebrate tissue concentrations that were required for modeling (Table D-6).

4 REFERENCES

Galbraith and MDEQ, 2004. Tittabawassee River Floodplain Screening-Level Ecological Risk Assessment- Polychlorinated Dibenzo-*p*-Dioxins and Polychlorinated Dibenzofurans. Galbraith Environmental Sciences and Michigan Department of Environmental Quality.

Integral, 2007. Human Health and Ecological Risk Assessment: St. Regis Paper Company Site, Cass Lake, MN. Unilateral Administrative Order Docket No. V-W-04-C-796. Prepared for International Paper, Memphis, TN. Prepared by Integral Consulting Inc. Mercer Island, WA. September 28, 2007.

Integral, 2010. Technical Memorandum on Bioaccumulation Modeling. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Prepared by Integral Consulting Inc., Seattle, WA. September 2010.

Kay, D., A. Blankenship, M. Zwiernik, J.L. Newsted, P.D. Jones, and J.P. Giesy, 2005. PCDDs and PCDFs in Aquatic and Terrestrial Food Webs of the Tittabawassee River, Michigan, USA. *Organohalogen Compounds* 67:2130-2133.

MACTEC, 2004. Interim Final Baseline Ecological Risk Assessment. Centredale Manor Restoration Project Superfund Site, Providence, Rhode Island. Submitted to U.S. Army Corps of Engineers, New England Division, under contract to Battelle. Contract No. DACW33-01-D-004. MACTEC Engineering and Consulting, Inc. September 2004.

Matscheko, N., M. Tysklind, C. De Wit, S. Bergek, R. Andersson, and U. Sellstrom, 2002. Application of sewage sludge to arable land-soil concentrations of polybrominated diphenyl ethers and polychlorinated dibenzo-*p*-dioxins, dibenzofurans, and biphenyls, and their accumulation in earthworms. *Environmental Toxicology and Chemistry* 21: 2515-2525.

Royall, R., 1986. The effect of sample size on the meaning of significance tests. *The American Statistician* 40(4):313-315.

Sample, B.E., J.J. Beauchamp, R.A. Efroymson, G.W. Suter II, and T.L. Ashwood, 1998. Development and Validation of Bioaccumulation models for Earthworms. ES/ER/TM-220. Prepared for the US Department of Energy, Office of Environmental Management. February 1998.

Sokal, R.R., and F.J. Rohlf, 1981. Biometry. Second Edition. W.H. Freeman and Co., San Francisco. 859pp.

Van den Berg, M., L.S. Birnbaum, M. Denison et al., 2006. The 2005 World Health Organization reevaluation of human and mammalian toxic equivalency factors for dioxins and dioxin-like compounds. *Toxicol. Sci.* 93(2):223-241.

TABLES

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-07[b]	2,3,7,8-TCDD	3.63E-07	U	9.86E-07	U
	1,2,3,7,8,9-HxCDD	6.41E-05		2.15E-05	
	1,2,3,4,7,8,9-HpCDF	3.58E-04		7.85E-06	J
	1,2,3,6,7,8-HxCDD	5.39E-04		4.87E-05	
	1,2,3,7,8-PCDD	1.12E-05		4.77E-06	JEMPC
	1,2,3,4,6,7,8-HpCDD	2.65E-02		6.63E-04	
	OCDD	4.74E-01	J	4.09E-03	
	2,3,7,8-TCDF	1.13E-05		8.49E-07	U
	1,2,3,7,8,9-HxCDF	7.34E-05		3.90E-06	U
	2,3,4,7,8-PCDF	7.72E-05		6.01E-06	J
	1,2,3,4,7,8-HxCDD	2.67E-07	U	1.03E-05	JEMPC
	1,2,3,6,7,8-HxCDF	7.10E-05		8.81E-06	J
	1,2,3,7,8-PCDF	5.25E-05	EMPC	1.15E-05	
	2,3,4,6,7,8-HxCDF	1.47E-04		9.83E-06	J
	1,2,3,4,6,7,8-HpCDF	3.19E-03		2.11E-04	
	1,2,3,4,7,8-HxCDF	3.67E-04		2.47E-05	
	OCDF	3.77E-02		3.54E-04	
ECO-08	2,3,7,8-TCDD	2.41E-07	U	6.45E-07	JEMPC
	1,2,3,7,8,9-HxCDD	1.24E-04	EMPC	8.75E-05	
	1,2,3,4,7,8,9-HpCDF	4.40E-04		2.53E-04	
	1,2,3,6,7,8-HxCDD	4.25E-04	EMPC	4.17E-04	
	1,2,3,7,8-PCDD	1.61E-05		2.16E-05	
	1,2,3,4,6,7,8-HpCDD	1.41E-02		6.18E-03	
	OCDD	2.07E-01	J	5.62E-02	J
	2,3,7,8-TCDF	4.10E-06		3.32E-06	
	1,2,3,7,8,9-HxCDF	2.97E-06	U	4.20E-06	U
	2,3,4,7,8-PCDF	4.88E-05		2.64E-05	
	1,2,3,4,7,8-HxCDD	2.63E-05	EMPC	5.15E-05	
	1,2,3,6,7,8-HxCDF	1.47E-04		8.49E-05	
	1,2,3,7,8-PCDF	6.20E-05		3.86E-05	
	2,3,4,6,7,8-HxCDF	2.19E-04		1.10E-04	
	1,2,3,4,6,7,8-HpCDF	3.92E-03		2.13E-03	
	1,2,3,4,7,8-HxCDF	3.95E-04		2.40E-04	
	OCDF	1.89E-02		5.06E-03	

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-09	2,3,7,8-TCDD	1.98E-07	U	1.38E-07	U
	1,2,3,7,8,9-HxCDD	1.62E-06	JEMPC	1.19E-05	
	1,2,3,4,7,8,9-HpCDF	6.95E-07	U	1.15E-05	
	1,2,3,6,7,8-HxCDD	4.30E-06		2.27E-05	
	1,2,3,7,8-PCDD	1.88E-07	U	2.21E-06	J
	1,2,3,4,6,7,8-HpCDD	1.79E-04		7.24E-04	
	OCDD	1.77E-03	J	6.67E-03	J
	2,3,7,8-TCDF	1.64E-07	U	1.12E-06	U
	1,2,3,7,8,9-HxCDF	2.05E-07	U	9.78E-08	U
	2,3,4,7,8-PCDF	1.77E-07	U	1.98E-06	J
	1,2,3,4,7,8-HxCDD	3.26E-07	U	5.27E-06	
	1,2,3,6,7,8-HxCDF	9.22E-07	JEMPC	5.02E-06	
	1,2,3,7,8-PCDF	1.82E-07	U	2.33E-06	J
	2,3,4,6,7,8-HxCDF	1.46E-06	J	6.94E-06	
	1,2,3,4,6,7,8-HpCDF	3.30E-05		1.40E-04	
	1,2,3,4,7,8-HxCDF	3.74E-06		1.38E-05	
	OCDF	1.08E-04		4.47E-04	
ECO-10	2,3,7,8-TCDD	8.70E-08	JEMPC	1.34E-07	U
	1,2,3,7,8,9-HxCDD	1.84E-06	J	2.28E-06	U
	1,2,3,4,7,8,9-HpCDF	1.38E-06	J	1.80E-06	U
	1,2,3,6,7,8-HxCDD	2.60E-06		3.89E-06	J
	1,2,3,7,8-PCDD	3.01E-07	J	5.88E-07	J
	1,2,3,4,6,7,8-HpCDD	8.44E-05		1.25E-04	
	OCDD	7.39E-04		1.31E-03	
	2,3,7,8-TCDF	2.73E-07	U	9.79E-07	U
	1,2,3,7,8,9-HxCDF	6.19E-07		2.11E-07	U
	2,3,4,7,8-PCDF	3.30E-07	JEMPC	4.33E-07	JEMPC
	1,2,3,4,7,8-HxCDD	6.40E-07	J	7.99E-07	J
	1,2,3,6,7,8-HxCDF	6.73E-07	J	1.01E-06	JEMPC
	1,2,3,7,8-PCDF	4.54E-07	J	6.75E-07	J
	2,3,4,6,7,8-HxCDF	9.45E-07	J	1.41E-06	U
	1,2,3,4,6,7,8-HpCDF	1.60E-05		2.90E-05	
	1,2,3,4,7,8-HxCDF	1.71E-06	J	2.70E-06	U
	OCDF	4.63E-05		7.68E-05	

Table D-1
Colocated Soil and Earthworm Data from the Cass Lake Superfund Site Used in the Development of Regression Relationships

StationID	Analyte	Concentration in Soil (mg/kg dw)	Soil Data Qualifier	Concentration in Earthworms (mg/kg dw[a])	Earthworm Data Qualifier
ECO-11	2,3,7,8-TCDD	2.27E-07	JEMPC	1.61E-07	U
	1,2,3,7,8,9-HxCDD	2.21E-05		8.65E-06	
	1,2,3,4,7,8,9-HpCDF	9.69E-06		2.87E-06	J
	1,2,3,6,7,8-HxCDD	3.46E-05		9.34E-06	
	1,2,3,7,8-PCDD	6.68E-06		1.74E-06	J
	1,2,3,4,6,7,8-HpCDD	5.44E-04		2.56E-04	
	OCDD	3.25E-03		1.81E-03	
	2,3,7,8-TCDF	3.38E-06		3.21E-06	U
	1,2,3,7,8,9-HxCDF	3.19E-06		1.01E-07	U
	2,3,4,7,8-PCDF	5.06E-06		1.09E-06	J
	1,2,3,4,7,8-HxCDD	1.19E-05		3.07E-06	JEMPC
	1,2,3,6,7,8-HxCDF	8.73E-07	J	1.76E-06	J
	1,2,3,7,8-PCDF	3.33E-06	EMPC	7.44E-07	J
	2,3,4,6,7,8-HxCDF	6.14E-06		1.34E-06	J
	1,2,3,4,6,7,8-HpCDF	9.50E-05		3.45E-05	
	1,2,3,4,7,8-HxCDF	1.34E-05		3.98E-06	J
	OCDF	3.17E-04		1.09E-04	
ECO-12[b]	2,3,7,8-TCDD	1.83E-06		1.28E-06	U
	1,2,3,7,8,9-HxCDD	2.35E-04		7.43E-05	
	1,2,3,4,7,8,9-HpCDF	8.68E-05		1.20E-04	
	1,2,3,6,7,8-HxCDD	1.94E-04		5.77E-04	
	1,2,3,7,8-PCDD	1.93E-05		1.69E-05	J
	1,2,3,4,6,7,8-HpCDD	7.44E-03		7.21E-03	
	OCDD	8.08E-02		4.37E-02	J
	2,3,7,8-TCDF	3.22E-06		2.88E-05	
	1,2,3,7,8,9-HxCDF	3.77E-05		5.01E-06	J
	2,3,4,7,8-PCDF	1.26E-05		8.32E-05	
	1,2,3,4,7,8-HxCDD	9.77E-05		2.72E-05	J
	1,2,3,6,7,8-HxCDF	2.16E-05		4.04E-05	
	1,2,3,7,8-PCDF	1.49E-05		5.13E-05	
	2,3,4,6,7,8-HxCDF	3.96E-05		3.61E-05	
	1,2,3,4,6,7,8-HpCDF	1.21E-03		1.19E-03	
	1,2,3,4,7,8-HxCDF	7.28E-05		1.61E-04	
	OCDF	4.25E-03		4.40E-03	

Notes

EMPC = estimated maximum possible concentration
J = estimated
U = not detected at the laboratory detection limit
a - Unless otherwise noted, earthworm tissue was collected at the same location as the adjacent soil data and has the same sample identification number.
b - Earthworm tissue is from a 28-d bioaccumulation test using Cass Lake soil.

Table D-2
Regression Relationships of Individual Dioxin and Furan Congeners in Colocated Cass Lake Soil and Earthworm Tissue

Congener	Relationship	Slope	Intercept	*p* value	R^2	Equation
2,3,7,8-TCDD	log-log	0.819	-2.494	**0.06**	0.53	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8-PCDD	log-log	0.516	-5.921	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,7,8-HxCDD	log-log	0.343	-7.481	0.2	0.16	ns
1,2,3,6,7,8-HxCDD	log-log	0.664	-3.420	**0.08**	0.48	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8,9-HxCDD	log-log	0.550	-5.043	**0.03**	0.65	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,6,7,8-HpCDD	log-log	0.479	-3.910	**0.1**	0.37	exp(intercept +slope*(ln($Cs_{congener}$))
OCDD	log-log	0.370	-3.424	0.2	0.26	ns
2,3,7,8-TCDF	log-log	0.251	-9.530	0.5	-0.12	ns
1,2,3,7,8-PCDF	log-log	0.593	-4.862	**0.08**	0.46	exp(intercept +slope*(ln($Cs_{congener}$))
2,3,4,7,8-PCDF	log-log	0.518	-5.916	0.2	0.29	ns
1,2,3,4,7,8-HxCDF	log-log	0.616	-4.292	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,6,7,8-HxCDF	log-log	0.609	-4.502	**0.03**	0.67	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,7,8,9-HxCDF	log-log	0.671	-5.742	**0.07**	0.52	exp(intercept +slope*(ln($Cs_{congener}$))
2,3,4,6,7,8-HxCDF	log-log	0.576	-5.218	**0.07**	0.49	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,6,7,8-HpCDF	log-log	0.593	-3.688	**0.05**	0.56	exp(intercept +slope*(ln($Cs_{congener}$))
1,2,3,4,7,8,9-HpCDF	log-log	0.453	-6.217	0.2	0.25	ns
OCDF	log-log	0.415	-4.735	0.2	0.28	ns

Notes

$Cs_{congener}$ = concentration of the given congener in soil

ns = no significant relationship

Regressions with $p \leq 0.1$ (in **bold)** are considered statistically significant and are used to construct regression equations.

Table D-3
Spearman's Correlation Coefficients (rho) for Dioxin and Furan Congeners in Cass Lake Earthworm Data For Which Regression Equations Could Not Be Developed

	1,2,3,7,8-PCDD	1,2,3,6,7,8-HxCDD	1,2,3,7,8,9-HxCDD	1,2,3,4,6,7,8-HpCDD	1,2,3,7,8-PCDF	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDF	1,2,3,4,6,7,8-HpCDF	2,3,4,6,7,8-HxCDF
1,2,3,4,7,8-HxCDD	**1.00**	0.96	**1.00**	0.93	0.96	**1.00**	**1.00**	0.82	**1.00**	0.96
OCDD	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93
2,3,7,8-TCDF	0.64	**0.71**	0.64	0.79	**0.71**	**0.64**	0.64	0.64	0.64	0.57
2,3,4,7,8-PCDF	0.96	**1.00**	0.96	0.96	**1.00**	0.96	0.96	0.86	0.96	0.93
1,2,3,4,7,8,9-HpCDF	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93
OCDF	**0.96**	0.93	**0.96**	**0.96**	0.93	**0.96**	**0.96**	0.71	**0.96**	0.93

Notes

Coefficients in **bold** are significant ($p < 0.05$, see Table D-4), highly correlated,and have a significant regression equation (per Table D-2).

Table D-4
Significance (*p-* values) of Spearman's Correlation Coefficients for Dioxin and Furan Congeners for Which Regression Equations Could Not Be Developed

	1,2,3,7,8-PCDD	1,2,3,6,7,8-HxCDD	1,2,3,7,8,9-HxCDD	1,2,3,4,6,7,8-HpCDD	1,2,3,7,8-PCDF	1,2,3,4,7,8-HxCDF	1,2,3,6,7,8-HxCDF	1,2,3,7,8,9-HxCDF	1,2,3,4,6,7,8-HpCDF	2,3,4,6,7,8-HxCDF
1,2,3,4,7,8-HxCDD	0	8.7E-07	0	4.9E-04	8.7E-07	0	0	4.1E-01	0	7.9E-10
OCDD	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04
2,3,7,8-TCDF	2.0E-03	7.0E-03	2.0E-03	7.0E-01	7.0E-03	2.0E-03	2.0E-03	3.7E-03	2.0E-03	3.4E-04
2,3,4,7,8-PCDF	8.7E-07	0	8.7E-07	6.4E-04	0	8.7E-07	8.7E-07	1.4E-01	8.7E-07	2.4E-05
1,2,3,4,7,8,9-HpCDF	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04
OCDF	7.6E-06	2.2E-03	7.6E-06	2.4E-07	2.2E-03	7.6E-06	7.6E-06	3.0E-01	7.6E-06	1.8E-04

Table D-5

Approach for Estimating Concentrations in Terrestrial Invertebrate Tissue of Congeners for Which a Significant Regression Equation Could Not Be Established

Congener of Interest	Related Congener	Average Ratio of Concentrations of the Congener of Interest to Related Congener Concentrations	CV
1,2,3,4,7,8-HxCDD	1,2,3,7,8-PCDD	1.94	0.221
	1,2,3,7,8,9-HxCDD	**0.430**	0.218
	1,2,3,4,7,8-HxCDF	0.375	0.575
	1,2,3,6,7,8-HxCDF	1.01	0.419
	1,2,3,4,6,7,8-HpCDF	0.0417	0.603
OCDD	1,2,3,7,8-PCDD	2,050	0.435
	1,2,3,7,8,9-HxCDD	460	0.443
	1,2,3,4,6,7,8-HpCDD	**8.02**	0.230
	1,2,3,4,7,8-HxCDF	349	0.407
	1,2,3,6,7,8-HxCDF	977	0.355
	1,2,3,4,6,7,8-HpCDF	38.0	0.340
2,3,7,8-TCDF	1,2,3,6,7,8-HxCDD[a]	**0.120**	1.179
	1,2,3,7,8-PCDF	1.16	1.398
	1,2,3,4,7,8-HxCDF[b]	**0.25**	1.228
2,3,4,7,8-PCDF	1,2,3,6,7,8-HxCDD	**0.108**	0.264
	1,2,3,7,8-PCDF	1.61	1.019
1,2,3,4,7,8,9-HpCDF	1,2,3,7,8-PCDD	5.06	0.768
	1,2,3,7,8,9-HxCDD	1.16	0.835
	1,2,3,4,6,7,8-HpCDD	0.0185	0.605
	1,2,3,4,6,7,8-HpCDF	0.0767	0.391
	1,2,3,4,7,8-HxCDF	**0.723**	0.332
	1,2,3,6,7,8-HxCDF	2.09	0.390
OCDF	1,2,3,7,8-PCDD	161	0.522
	1,2,3,7,8,9-HxCDD	36.2	0.546
	1,2,3,4,6,7,8-HpCDD	**0.603**	0.215
	1,2,3,4,7,8-HxCDF	25.1	0.257
	1,2,3,6,7,8-HxCDF	72.6	0.334
	1,2,3,4,6,7,8-HpCDF	2.79	0.256

Notes

CV = Coefficient of variation, a measure of dispersion of the data, which is calculated as the standard deviation of the ratios of the congener to its correlate divided by the average ratio.

Selected average ratio for estimating tissue concentrations for the congener of interest are in **bold**.

a - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples outside of the impoundments.

b - Selected congener for estimating 2,3,7,8-TCDF tissue concentrations from soil samples inside the impoundments.

Table D-6
Estimated Concentrations of Dioxins and Furans (TEQ_{DF}) in Terrestrial Invertebrate Tissue at the Site

	CT, mg/kg dw	RM, mg/kg dw
With RBA[a]		
$TEQ_{DF,B}$[b]	6.07E-05	1.81E-04
$TEQ_{DF,M}$[c]	9.42E-05	2.84E-04
Without RBA[a]		
$TEQ_{DF,B}$[b]	1.17E-04	3.59E-04

Notes

CT = central tendency

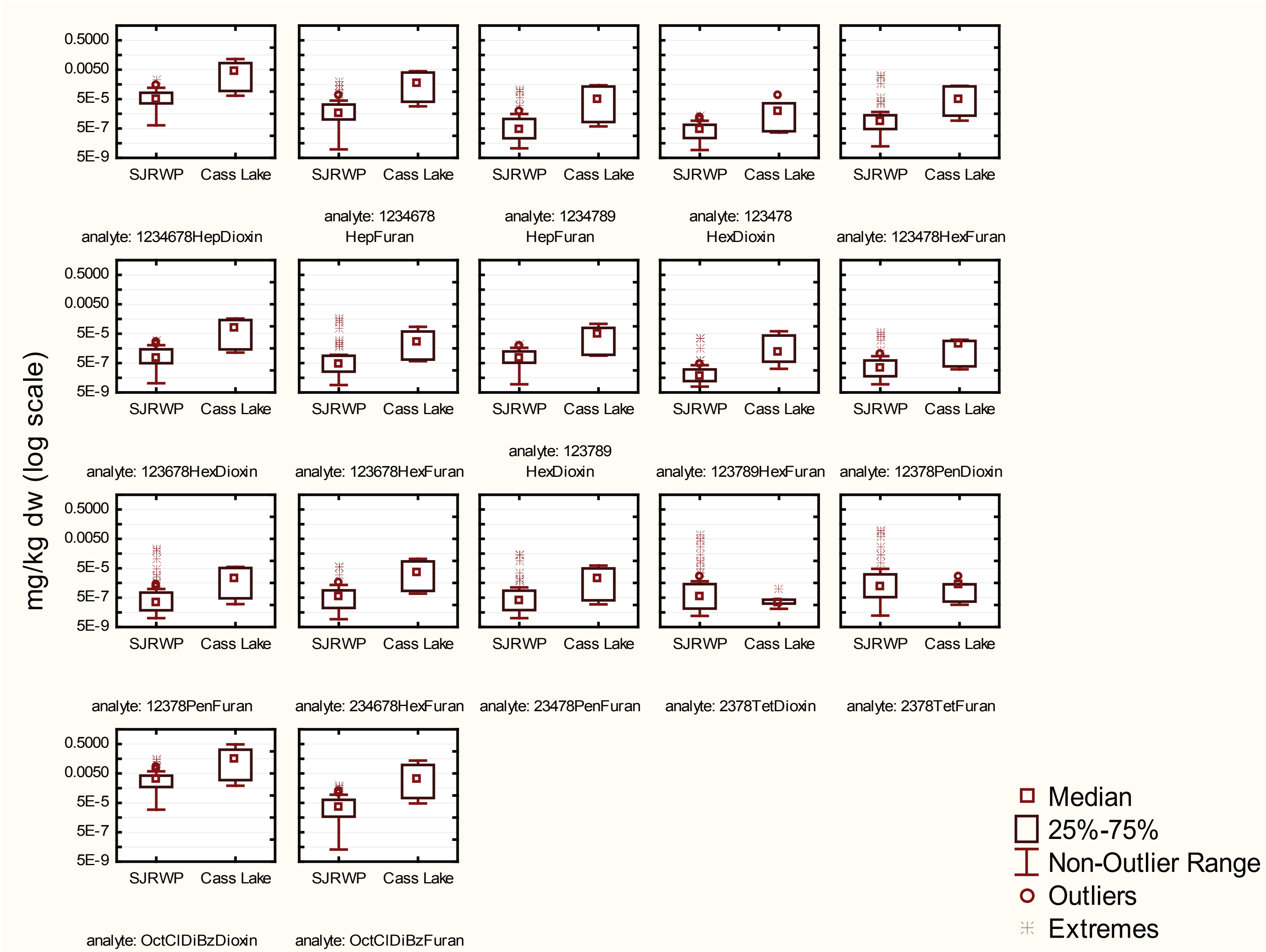
RM = reasonable maximum

a - Relative bioavailability adjustment factor applied to TCDD congener for calculating $TEQ_{DF,B}$

b - Calculated using soils north of I-10, consistent with the killdeer exposure scenario

c - Calculated using peninsula-wide soils, consistent with the raccoon exposure scenario

FIGURES



ANCHOR QEA integral consulting inc.

Figure D-1
Box Plots of Dioxin and Furan Congener Concentrations in Cass Lake and all SJRWP Soils from North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

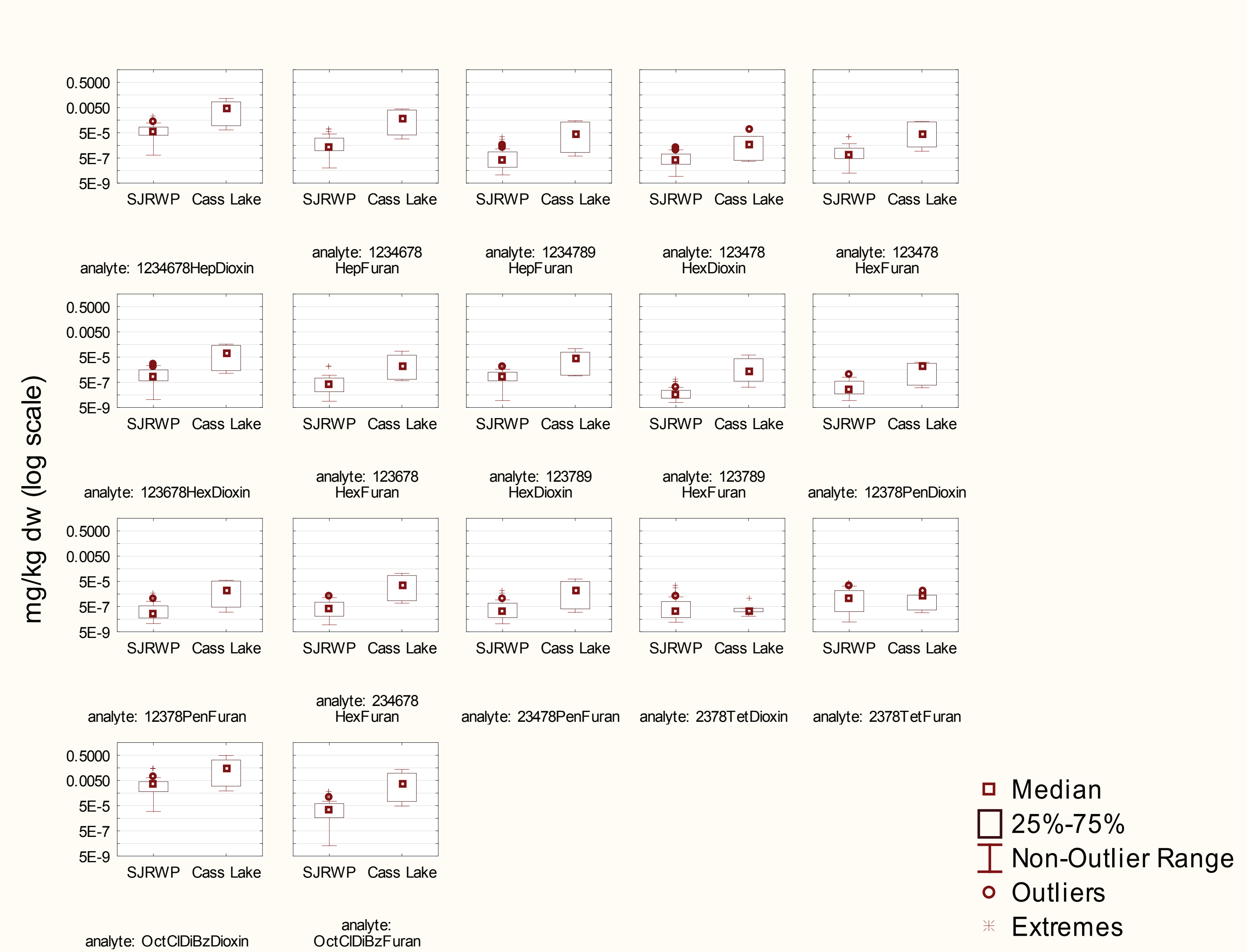
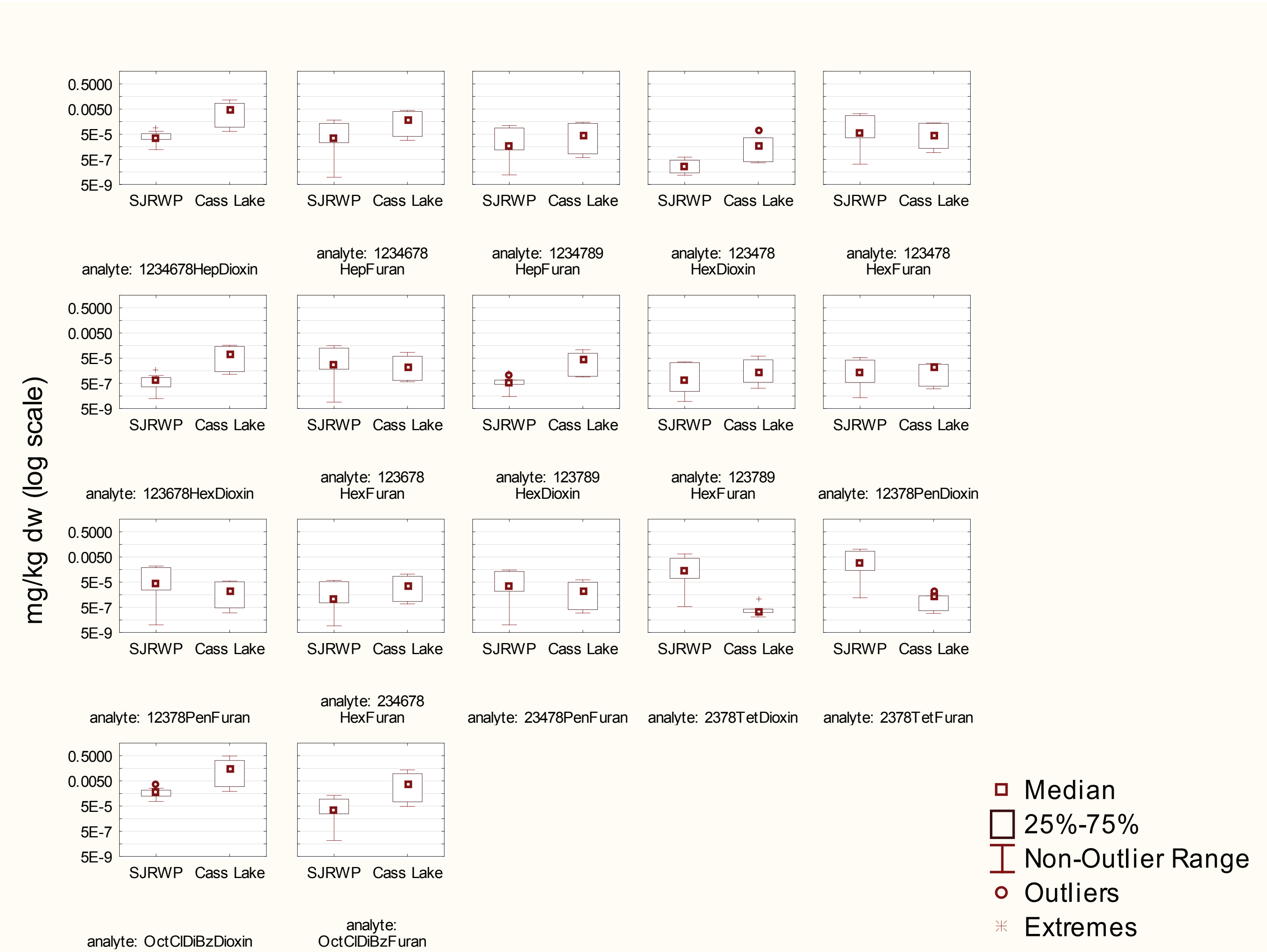


Figure D-2
Box Plots of Dioxin and Furan Congener Concentrations in
Cass Lake and SJRWP Soils Outside of the Waste Impoundments North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC



ANCHOR QEA integral consulting inc.

Figure D-3
Box Plots of Dioxin and Furan Congener Concentrations in
Cass Lake and SJRWP Soils Inside of the Waste Impoundments North of I-10
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

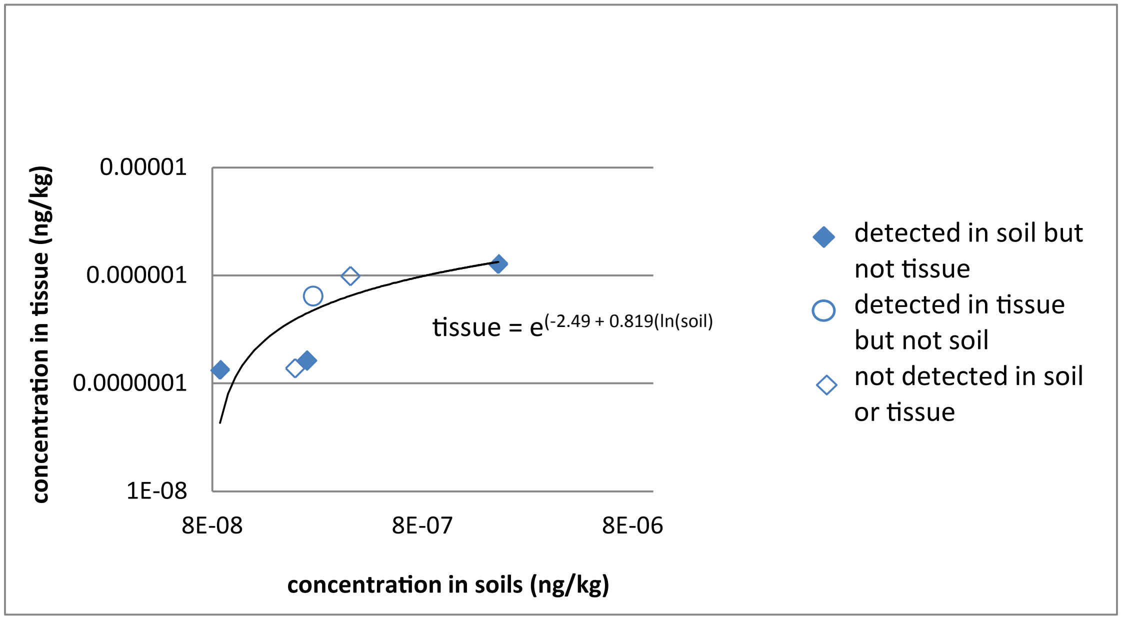


Figure D-4
Relationship of 2,3,7,8-TCDD in Cass Lake Colocated Soil and Earthworm Tissue Samples
SJRWP Baseline Ecological Risk Assessment
SJRWP Superfund/MIMC and IPC

APPENDIX E
SCREENING-LEVEL
ECOLOGICAL RISK ASSESSMENT,
SOUTH IMPOUNDMENT

SCREENING-LEVEL ECOLOGICAL RISK ASSESSMENT
SOUTH IMPOUNDMENT
SAN JACINTO RIVER WASTE PITS SUPERFUND SITE

Prepared for

International Paper Company

U.S. Environmental Protection Agency, Region 6

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August 2012

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LIST OF ACRONYMS AND ABBREVIATIONS

Abbreviation	Definition
BERA	baseline ecological risk assessment
CDD	chlorinated dibenzo-*p*-dioxins
COI	chemical of interest
COPC	chemical of potential concern
$COPC_E$	chemical of potential ecological concern
CSM	conceptual site model
DQO	data quality objective
EcoSSL	ecological soil screening level
ERA	ecological risk assessment
I-10	Interstate Highway 10
IPC	International Paper Company
K_{ow}	octanol-water partition coefficient
PCB	polychlorinated biphenyl
RI/FS	Remedial Investigation and Feasibility Study
SAP	sampling and analysis plan
Site	San Jacinto River Waste Pits Site
SLERA	screening-level ecological risk assessment
SLV	screening level value
SMDP	scientific management decision point
SVOC	semivolatile organic compound
TCEQ	Texas Commission on Environmental Quality
USEPA	U.S. Environmental Protection Agency

1 INTRODUCTION

This Screening-Level Ecological Risk Assessment (SLERA) for the south impoundment area of the San Jacinto River Waste Pits site (the Site) has been prepared on behalf of International Paper Company (IPC), pursuant to the requirements of Unilateral Administrative Order, Docket No. 06-03-10, which was issued by the U.S. Environmental Protection Agency (USEPA) to IPC and McGinnes Industrial Maintenance Corporation on November 20, 2009 (USEPA 2009). This SLERA presents information to supplement the SLERA prepared as Appendix B to the Remedial Investigation/Feasibility Study (RI/FS) Work Plan (Anchor QEA and Integral 2010).

This document is submitted as Appendix E to the Baseline Ecological Risk Assessment (BERA), and uses results from the Phase I soil investigation conducted in March 2011 in Soil Investigation Area 4 (Integral 2011b). Results of the Phase I investigation for this area have been presented in the Preliminary Site Characterization Report (PSCR) (Integral and Anchor QEA 2012). Analyses of the data according to the data quality objectives (DQOs) of the Phase I study are presented in Attachment A to the draft Soil Sampling and Analysis Plan (SAP) Addendum 3 (Integral 2011a). This SLERA was prepared consistent with USEPA guidance for ecological risk assessment (USEPA 1998) and addresses Step 1 and Step 2 of the 8-step ecological risk assessment (ERA) process for Superfund (USEPA 1997) (Figure E-1). Preparation of a SLERA for the south impoundment provides the basis for a BERA for the south impoundment, which will be submitted with the Remedial Investigation Report. The south impoundment BERA will be conducted using the approach and methods for completing ERA Steps 3 through 8 as described in the RI/FS Work Plan.

This SLERA provides the screening-level problem formulation and ecological effects evaluation (Step 1) and the screening-level exposure assessment and risk evaluation (Step 2), either as a unique section in this document, or by reference to Appendix B of the RI/FS Work Plan. Several components of the SLERA are addressed only briefly in this document because they are described in greater detail elsewhere:

- Site history and facilities used at the Site (Section 2 of the RI/FS Work Plan and Section 1.4.1 of Soil Sampling and Analysis Plan (SAP) Addendum 1 (Integral 2011b)
- Identification of chemicals of interest (COIs) (Appendix C to the RI/FS Work Plan)

- Complete evaluation of Phase I soil investigation results according to DQOs in Attachment A to Soil SAP Addendum 3 (Integral 2011a).

This SLERA is intended to provide a description of the environmental setting of Soil Investigation Area 4, and to document the scientific management decision points (SMDPs) to transition from a general understanding of the south impoundment environment to the more Site-specific study design elements and analyses required for performing the BERA. The SLERA is organized as follows:

- **Section 2**. Screening-Level Problem Formulation. This section reviews the relevant information for the south impoundment area. It includes by reference related information from the overall SLERA for the Site. The result is a list of receptor surrogates and an ecological conceptual site model (CSM) for the south impoundment area.
- **Section 3**. Screening-Level Evaluation and Identification of Chemicals of Potential Ecological Concern (COPCES). This section describes the basis for the screening-level values (SLVs) used in the risk-based screens, and presents a screening analysis of data collected during the initial soil investigation described by Soil SAP Addendum 1 (Integral 2011b), and results in the identification of COPCES.
- **Section 4**. Scientific Management Decision Point (SMDP). A summary of the findings of the SLERA is presented as an SMDP for the south impoundment ERA process. Uncertainties in the analyses are discussed.

2 SCREENING LEVEL PROBLEM FORMULATION

The screening level problem formulation uses existing information to develop a preliminary CSM for ecological receptors that addresses the following (USEPA 1997):

- The environmental setting and contaminants known or suspected to occur
- Mechanisms of contaminant fate and transfer
- Mechanisms of toxicity and likely categories of receptors that could be affected
- The complete exposure pathways linking contaminants to ecological receptors in the south impoundment environment
- Endpoints that can be used to screen for potential ecological risk.

This section summarizes basic information on the environmental setting, chemical fate and transport mechanisms relevant to developing the CSM, receptors potentially in the south impoundment area of the Site, and the surrogates to be used for risk assessment, and it defines the assessment endpoints for the screening level analysis. A detailed discussion of the toxicity of dioxins and furans is provided in Attachment B2 to Appendix B of the RI/FS Work Plan. The resulting CSM synthesizes this preliminary assessment to identify mechanisms of exposure and effects that may result in contaminant-related risks to ecological receptors. The CSM may be refined in the problem formulation presented in the BERA to better reflect site-specific exposures and risks.

2.1 Environmental Setting

Several historical aerial images of this area of the Site were analyzed to determine the location and history of the impoundment south of Interstate 10 (I-10). Review of historical documents and aerial photographs indicates that an impoundment (Figure E-2) was constructed in the mid-1960s south of I-10, on the peninsula directly south of the northern impoundments on the western bank of the San Jacinto River, in Harris County, Texas (this area is Soil Investigation Area 4 for the purposes of the RI/FS). This southern impoundment was used for disposal of paper mill wastes. Beginning in the 1970s, much of the peninsula south of I-10 underwent substantial physical change due to road, parking lot, and building development. Additional description of the possible configurations of the original southern impoundment, documented historical waste disposal and characteristics of that waste, and physical changes at the Site from 1957 to the present are provided by Integral (2011b).

The upland areas south of I-10 are currently under industrial or commercial use, including use by a shipbuilding company and an active shipyard. The shipyard has been operational since 1957, and the shipyard property was the site of a waste impoundment used for management of wastes associated with barge repair and cleaning materials (e.g., grinding or blasting wastes and cleaning solutions). A recent application for a municipal settings designation by the shipyards (W&M 2011) describes an ongoing groundwater treatment program, and the history of groundwater monitoring and remediation on the shipyards property, which is to the west of Soil Investigation Area 4.

The PSCR (Integral and Anchor QEA 2012) provides a more complete summary of recent information on commercial and industrial activities in the vicinity of the south impoundment, and uses observations made during the Phase I soil investigation to update the south impoundment CSM that was developed in Soil SAP Addendum 1 (Integral 2011b). The most recent CSM, from the final PSCR, is presented in greater detail in Section 2.6.

2.1.1 Habitat

In natural low-elevation habitats adjacent to the San Jacinto River, soils consist primarily of clay and sand and support loblolly pine-sweetgum, loblolly pine-shortleaf pine, water oak-elm, pecan-elm, and willow oak-blackgum (TSHA 2009). The area of the south impoundment has been cleared and graded for industrial use. This area is generally flat with very little topographic relief, and a small rise in elevation from 6 feet above sea level in the north to greater than 10 feet above sea level at the southern end of the impoundment. Most of the area is covered with mowed grasses and forbs and there are few trees throughout the Site. Shorelines in the area are covered by rip-rap, and fringe wetlands are populated by phragmites, small trees, and shrubs, particularly along the northern boundary of the south impoundment. Shallow estuarine waters abut the shoreline, and deeper estuarine waters offshore to the west of the Site are maintained for shipping activities.

2.1.2 Contaminants Known or Suspected to Occur at the Site

The process to identify COIs (Table E-1) for the Site is described in Appendix C of the RI/FS Work Plan (Anchor QEA and Integral 2010). COIs were defined as those chemicals that are among USEPA's priority pollutants, were reported by one or more technical papers as potentially occurring in pulp mill solid wastes or leachate from solid waste landfills containing pulp mill wastes, and are likely to have bound to organic carbon or could otherwise have persisted for more than 40 years in the Site environment. The Phase I soil investigation for the Site generated information on the spatial distribution and concentrations of COIs in soils of the south impoundment area (Integral 2011a).

$COPC_{ES}$ for the south impoundment area have not been identified. The methods to determine $COPC_{ES}$, and the analyses of Phase I soil data to identify $COPC_{ES}$ for evaluation in the BERA is presented below, in Section 3.

2.2 Contaminant Fate and Transport

The characterization of contaminant fate and transport includes identification of 1) pathways for migration of COIs at the Site; and 2) physical, chemical, and biological transformations of these COIs. Understanding general mechanisms of fate and transport helps define the exposure pathways to ecological receptors that may be adversely affected by Site contaminants (USEPA 1998). This section provides general information on chemical transport and transformation pathways for COIs in soils of the south impoundment on the basis of currently available information. The PSCR provides a more detailed discussion.

2.2.1 Physical Fate and Transport Processes

Potential transport pathways for COIs in south impoundment soils include transport of contaminated soils via surface runoff, and transport of dissolved COIs via groundwater to surface water. The updated CSM for the south impoundment presented in the PSCR (Figure E-3) differentiates surface soil and subsurface soil on the basis of information developed in the Phase I soil investigation. If surface soils in the south impoundment area are affected by paper mill wastes, and surface water runoff pathways in the area could transport soils to the Old River, sediment and water could be affected by COIs originating in the south impoundment.

To evaluate the potential for physical transport of COIs from the surface soils to the aquatic environment, the topography and surface water flow paths south of I-10 were described in the PSCR (Section 7.1). Surface water flow paths were shown to comingle into larger drainage networks, but to ultimately either discharge to the river on either side of the peninsula or terminate in surface depressions, at which surface water runoff would be expected to aggregate and ultimately percolate into the soil. Given that dioxin and furan concentrations in the three sediment samples nearest the south impoundment are below the reference envelope value for soil, and that the concentrations of dioxins and furans in the majority of surface soil samples were also below concentrations in background soils, the potential for surface transport of soils affected by paper mill wastes to the aquatic environment is considered negligible (Integral and Anchor QEA 2012). USEPA has requested that additional information on sediments and soils be collected as part of a subsequent phase of investigation.

The subsurface soils 6 feet and deeper below ground surface may be in contact with groundwater, and chemicals could be transported in a dissolved state or bound to particulates from the subsurface soil environment to the aquatic environment. The groundwater study conducted north of I-10 (described in Section 6 of the PSCR) is relevant to the CSM for the south impoundment because it demonstrated that, even in an area where there are concentrated wastes situated in alluvial sediments, contamination of alluvial groundwater and the deep aquifer with dioxins and furans did not occur. These results suggest that in the vicinity of the south impoundment, where the data indicate that paper mill wastes are substantially less concentrated than in the location of the groundwater study, that there may also be a very limited or no groundwater pathway resulting in the transport of dioxins and furans to receptors. Also, dioxins and furans strongly adsorb to soil particles and have very low solubility and mobility in groundwater (Fan et al. 2006; USAF 2006; ATSDR 1998). ATSDR (1998) indicates that chlorinated dibenzo-*p*-dioxins (CDDs) "…bind strongly to the soil, and therefore are not likely to contaminate groundwater…" and "CDDs are unlikely to leach to underlying groundwater…" These properties further decrease the likelihood that dioxins and furans are transported by groundwater from the subsurface soils to the aquatic environment.

However, USEPA contends that additional groundwater data are necessary to demonstrate that conditions observed north of I-10 are representative of those south of I-10, and has requested that information on the chemistry of alluvial groundwater in the area south of I-10 be collected. If there is a groundwater pathway resulting in contamination of the aquatic environment with chemicals from the southern impoundment, analysis of the sediment and tissue data for the overall Site in the BERA will address any related ecological risks.

2.2.2 Biological Fate and Transport Processes: Bioaccumulation

Bioaccumulation is relevant to the BERA for several chemicals. A simple definition of bioaccumulation is the sequestration of a chemical substance in an organism when the absorption rate (from exposures to all media) exceeds the elimination or transformation rates, resulting in the concentration in tissue exceeding the concentration in the exposure medium. Bioaccumulation dynamics and rates are specific to the substance of concern, the exposure route, the medium or media in which the chemical is delivered, and the type of organism. Biomagnification is related to bioaccumulation and describes the increase in the concentration of a substance with increasing trophic level in a food chain (e.g., from primary to tertiary consumer). Biomagnification appears to be restricted to a relatively small group of chemicals (Croteau et al. 2005; Suedel et al. 1994).

A key indicator of the potential for bioaccumulation is the chemical's hydrophobicity, which is most often expressed using the *n*-octanol–water partition coefficient, K_{ow}, and has been used to predict bioaccumulation potential. Hydrophobic and lipophilic organic compounds that are resistant to both degradation and excretion in organisms build up in adipose tissue. Generally, organic chemicals that significantly bioaccumulate are those that are non-ionic, have a log K_{ow} of 5 or greater, and are not rapidly metabolized or excreted (USEPA 2008a). More recent literature indicates that dioxin and furan congeners that are not tetrachlorinated at the 2, 3, 7, and 8 positions have very limited bioaccumulation potential in vertebrates (USEPA 2008b). The bioaccumulation of dioxins and furans is addressed by Integral (2010).

Metals bioaccumulation is complex, and bioaccumulation rates can vary with the concentration in the exposure medium. As a result, simple models of metals bioaccumulation, such as the use of bioaccumulation factors, may lead to inaccurate depiction of concentrations in tissue (USEPA 2007). Bioaccumulation is considered a relevant process for determining the fate of COIs at the Site, and chemical-specific bioaccumulation potential based on the Texas Commission on Environmental Quality

(TCEQ) guidance (TCEQ 2006) is incorporated into the risk-based screens that are applied in Appendix C (to the RI/FS Work Plan) to identify chemicals of potential concern (COPCs) for ecological (and human) receptors. The findings of the Technical Memorandum on Bioaccumulation Modeling (Integral 2011d) including the conceptual framework on bioaccumulation of dioxins and furans, and the general conclusions regarding the appropriate models for predicting tissue concentrations, and the chemicals considered bioaccumulative by TCEQ (2006) provide the basis for consideration of trophic transfer to ecological receptors for this risk assessment.

2.3 Selection of Surrogate Ecological Receptors

This section builds from Appendix B and Attachment B1 to the RI/FS Work Plan (Anchor QEA and Integral 2010) to describe the ecological receptors that could occur in the vicinity of the south impoundment at the Site, identify surrogate species to be evaluated in the BERA for the south impoundment and present the rationale for their selection. A surrogate receptor species is chosen to represent a group of related species with similar feeding patterns, habitat associations, or other life history characteristics that affect the exposure potential of the receptor group.

2.3.1 Selected Receptor Surrogates

Ecological receptor surrogates are considered representative of the trophic and ecological relationships known or expected at the Site. In selecting receptor surrogates for evaluation in the BERA for the Site, the following criteria were considered:

- The receptor is or could potentially be present at the Site
- The receptor is representative of one or more feeding guilds
- The receptor is known to be either sensitive or potentially highly exposed to COPCs at the Site
- Life history information is available in the literature or is available for a similar species that can be used to inform life history parameters for the receptor.

Detailed tables listing the species of plants, reptiles, birds, and mammals that could use the upland habitats on the Site or in the vicinity of the Site are provided in the SLERA for the overall Site, as Attachment B1 to the RI/FS Work Plan (Anchor QEA and Integral 2010).

Using the guidelines listed above, four upland receptors (a bird, two mammals, and a reptile) were selected for evaluation in the BERA for the southern impoundment (Table E-2). Additional information on these receptors' life history and feeding behavior is provided below.

2.3.1.1 Reptile—Common Garter Snake

The garter snake was selected because it is a common, invertivorous reptile whose habitat requirements overlap with the conditions present in the upland portions of the Site.

The common garter snake (*Thamnophis sirtalis*) is one of the most abundant snakes in North America. Of the four subspecies of the common garter snake found in Texas, the Texas garter snake (*Thamnophis sirtalis annectens)* is the only subspecies known to inhabit eastern Texas locations; Harris County is one of several upper Gulf Coast counties in which these snakes have been observed in the last decade (Cannatella and LaDuc 2011). Regional populations of common garter snakes across the continent are distinguished mostly by variation in color patterns. The adult common garter snakes range in size between 46 and 137 cm (18 and 54 inches), and weigh an average of 150 g. The males are smaller than females and the young, which are similar in appearance to the adults, are 12.5 to 23 cm (5 to 9 inches) long at birth (Zimmerman 2002).

The adaptability and resilience of the common garter snakes are evidenced by their residence in a wide variety of terrestrial and semiaquatic habitats, including meadows, marshes, woodlands, hillsides, and suburban and urban areas where debris, rock walls, foundations, gardens and other features provide good cover. These snakes prefer moist, grassy environments such as is found near the edges of ditches, ponds, lakes and streams (Zimmerman 2002). In Texas, these snakes are found primarily in lowland habitats, particularly in areas with standing or running water, but can also be seen in open or edge habitats (Cannatella and LaDuc 2011). Similar to most reptiles, the common garter snake uses thermoregulation to achieve a preferred body temperature between 28 and 32°C. While these snakes tolerate a broader range of temperatures than do most, they bask in the sun during the day, and convene in coiled masses during sleep or hibernation to retain body heat. Hibernation occurs in natural cavities, rodent or crayfish burrows, under rock piles, or in stumps.

The common garter snake eats a variety of prey, dependent primarily on whether it is appropriately sized for swallowing whole. The adult diet includes amphibians, fish, and insects. Juvenile garter snakes eat a greater proportion of earthworms and insects than do adults. Baby birds, mammals, molluscs, and other snakes are also taken as prey items (Cannatella and LaDuc 2011)

Garter snakes mate in the spring, as soon as they emerge from hibernation, and are ovoviviparous, meaning they carry their young until birth. In the summer and fall, the females birth an average of 26 young. The mother snakes allow the young to be around them for several days after birth, but do not provide any care, protection, or nourishment. These snakes reach sexual maturity, and maximum size, at 3 to 4 years of age, though Zimmerman (2002) indicates that the average lifespan of common garter snakes is approximately 2 years and that most common garter snakes probably die in their first year of life.

Common garter snakes are eaten by a wide variety of predators, including large fish, bullfrogs, snapping turtles, milk snakes, American crows, hawks, great blue herons, raccoons, foxes, squirrels, and shrews.

2.3.1.2 Bird—Killdeer

The killdeer (*Charadrius vociferous*) was selected because it is an upland bird whose habitat requirements overlap with the conditions present in the upland portions of the site. Only terrestrial birds are expected to be present in the south impoundment area.

The killdeer is an upland plover that feeds mainly on terrestrial invertebrates (e.g., earthworms, beetles, grasshoppers, and other small invertebrates). Stomach contents from killdeer in Texas were reported to contain 98 percent animal matter, mostly worms and insects (McAtee and Beal 1924). The species is widespread throughout North America, using open area habitats (e.g., agricultural fields, lawns, golf courses). The killdeer is non-migratory within its range in the southern United States, including Texas (Jackson and Jackson 2000). It is known to be common year-round in the vicinity of the Site (Attachment B1 to the RI/FS work plan [Anchor QEA and Integral 2010]). This species is tolerant of constructed disturbances, and nesting has been documented to occur in construction sites, road shoulders, and graveled rooftops (Jackson and Jackson 2000). Average nesting home ranges of killdeers in Minnesota were relatively small (0.57 acres). Larger, year-round home ranges of approximately 15 acres are reported elsewhere; nesting period home ranges are smaller. Nesting in Mississippi occurs from mid-March through late July and involves multiple broods (Jackson and Jackson 2000).

Due to its likely presence in the upland portions of Site, its relatively small home range and site fidelity, and its predominantly terrestrial invertebrate diet, the killdeer is representative of the species that would be subject to ecological risks associated with the terrestrial food chain at the Site. The use of this surrogate species is considered protective of smaller home range bird species at the Site (e.g., sparrows, wrens) that likely eat a larger percentage of plant matter, as well as larger omnivores (e.g., crows), and would also be protective of terrestrial carnivores (e.g., hawks) that likely have larger home and forage ranges.

2.3.1.3 Mammal—Pocket Gopher

The Baird's pocket gopher (*Geomys breviceps*), also known as the Louisiana pocket gopher, is virtually indistinguishable, morphologically, from the plains (*G. busarius*) and Attwater's (*G. attwateri)* pocket gophers, each of which inhabit different regions of Texas (Sulentich et al. 1991; Davis and Schmidly 1994). These pocket gophers are small, dark brown, burrowing herbivores. With long, curved and specially adapted front claws, a broad, flat head, tiny, bead-like eyes and rudimentary ears, and a compact body with skin and hair arranged to allow movement through borrows both backward and forward, these gophers are more highly specialized for digging than any other North American rodent (Davis and Schmidly 1994; KSR 2011; Sulentich et al. 1991). *G. breviceps* is the smallest of its congenerics, averaging 208 mm in length and weighing between 78 and 150 g, with an average reported weight of 100 g (MNH 2012). The Baird's pocket gopher is found in the eastern portion of Texas and has been found on both sides of the San Jacinto River in Harris County (Sulentich et al. 1991; Davis and Schmidly 1994).

Geomys live underground most of their lives and maintain labyrinths of burrows in sandy and loamy soils, digging to an average depth of approximately 6 inches and up to 2 feet, generally on treeless land (Davis and Schmidly 1994). Because much of the burrowing is done in search of food, tunnels meander through feeding areas, and can extend well over 100 m. These rodents are solitary; each tunnel system is occupied by only one gopher. They rarely leave their burrows, except at night for mating or for limited foraging beyond the entrance (KSR 2011). In wet months, pocket gophers are known to live and nest in above-ground mounds of dirt, in order to avoid being flooded out of their burrows and tunnels (Sulentich et al. 1991).

The Baird's pocket gopher is an herbivore, obtaining most of its food while digging tunnels and feeding primarily on underground roots and the stems of weeds and grasses. Although most plant food is encountered and ingested while the gopher digs its lateral tunnels, green plants and grasses are obtained at night from around the entrance of the tunnels and beyond. Fur-lined cheek pouches are used to carry food and nesting material. Cellulose-digesting bacteria in the digestive system help the Baird's pocket gopher digest grasses and stored underground rhizomes during the winter and these gophers, as do many rodents, increase their utilization of food by ingesting their fecal pellets (Sulentich et al. 1991; Davis and Schmidly 1994).

The Baird's pocket gopher begins breeding in eastern Texas in early February and continues through August, with peak productivity occurring in June and July. One to four young are born to each litter (Sulentich et al. 1991). The young remain with their mother until nearly full-grown, at about 6 to 7 weeks of age, when they disperse to lead an independent life (Davis and Schmidly 1994). Sexual maturity is reached within 90 days of birth (Sulentich et al. 1991).

In east Texas, Baird's pocket gophers are preyed on by long-tailed weasels, and, when caught out of their burrows, are vulnerable to king snakes, great-horned owls, red-tailed hawks, and striped skunks, among other common rodent predators (Sulentich et al. 1991; Davis and Schmidly 1994). Because they remain protected in their borrows most of the time, pocket gophers are long-lived relative to many other rodents, living an average of 1 to 2 years in the wild (Davis and Schmidly 1994). The estimated population density in prairie habitat near College Station, Texas, was approximately 0.55 gophers per hectare (Sulentich et al. 1991).

2.3.1.4 Mammal–Virginia Opossum

The Virginia opossum (*Didelphis virginiana*) was selected because it is an omnivorous mammal whose habitat requirements overlap with the conditions present in the upland portions of the site.

The Virginia opossum is a widespread and adaptable nocturnal scavenger similar in size to a large house cat (UMMZ 2003). It is the only marsupial found north of Mexico. Opossums range from Central America through much of the continental United States, including the eastern two-thirds of the country and the coastal Pacific. Opossums range in size from 350 to 940 mm, averaging 740 mm. Adult males weigh an average of 5.5 pounds, and adult females average 4.0 pounds (Georgia DNR 2012); size may vary with location and climate (MNH 2012). The lifespan of a Virginia opossum averages 2 years, though many die in the first year of life (TPWD 2012). Both northern and southern populations have white fur with black tips. They have a pointed snout, opposable thumb-like appendages and a scaly prehensile tail that can be used to climb, hang, or grasp objects (TPWD 2012).

Opossums are well adapted to living near humans and occur in a variety of habitat types. They are primarily found in woodland areas especially near creeks, rivers, or lakes, but can also occupy marshes, farmland, prairies, and urban and rural environments. They prefer to live in hollow trees and logs, but can also nest under rocks, buildings, bridges, attics, woodpiles or in other animals' abandoned burrows. (UMMZ 2003; Georgia DNR 2012). In East Texas, Virginia opossums typically frequent overlapping home ranges approximately 0.05 km^2 in size, although the minimum size of home ranges may vary from 0.001 to 0.23 km^2. In East Texas woodland habitat, the density of opossums is about one opossum every 0.02 km^2 while in sandy, coastal parts of the state, the density is about one opossum every 0.06 km^2 (Davis and Schmidly 1994).

The Virginia opossum has a brief gestation period of 2 weeks after which the relatively undeveloped young crawl from the birth canal and attach themselves to the mother's nipple inside of her fur-lined pouch, where they stay attached for 7 weeks of nursing (UMMZ 2003). Litters usually consist of seven young, and Virginia opossums typically have two litters per year (Georgia DNR 2012).

Virginia opossums are omnivorous. Consuming mostly insects and carrion, the opossum also forages for acorns, berries, and other fruit and is also known to eat crustaceans, frogs, bird eggs and nestlings, small rodents, and the young of its own kind. In human-populated areas the opossum is known to scavenge for garbage and can be considered a nuisance for this reason (Georgia DNR 2012).

Common predators of Virginia opossums include canids, raccoons, and raptors. Humans are also a main cause of mortality through hunting and trapping, and opossums are frequently killed on roads (Georgia DNR 2012). Opossums are considered a game animal and in many states there are rules and regulations pertaining to their harvest through trapping and hunting. Despite their appeal to hunters, biologists do not believe that hunting is a threat to most populations of this species (Georgia DNR 2012).

2.3.2 Threatened and Endangered Species

Attachment B1 to the RI/FS work plan (Anchor QEA and Integral 2010) provides lists of species that could occur at the Site. Among the animals listed in Attachment B1 to the RI/FS work plan (Anchor QEA and Integral 2010), the upland species that are state-listed as threatened or endangered are:

- Timber rattlesnake
- Smooth green snake
- Rafinesque's big-eared bat.

The two snakes that are listed above have habitat requirements that are inconsistent with conditions present on the site (Anchor QEA and Integral 2010). The common garter snake has been selected as the surrogate receptor for reptiles at the Site. The Rafinesque's big-eared bat is not expected to use the habitats found in the vicinity of the Site because it feeds primarily on emergent aquatic insects, which are generally restricted to freshwater systems and are uncommon in brackish estuarine waters found near the Site.

In addition to these listed species, the American bald eagle, protected under the federal Bald and Golden Eagle Protection Act and listed as threatened by the State of Texas may be found in the vicinity. The American bald eagle may hunt for fish or eat carrion found on terrestrial and shoreline areas. Given the limited size and habitat south of I-10, the bald eagle is considered unlikely to occur and is not addressed specifically for Soil Investigation Area 4.

2.4 Potential Routes of Exposure

For an exposure pathway to be complete, a contaminant must be able to travel from its source to an ecological receptor, and to be taken up by the receptor by one or more exposure routes. Complete exposure pathways for terrestrial wildlife result from ingestion of contaminated soil; ingestion of prey organisms that have been exposed to contaminated media and have bioaccumulated COIs; direct contact with contaminated soil; and inhalation of volatile chemicals in confined spaces (burrows). Interpretation of the significance of each exposure route in any species is dependent upon the availability of information in the literature. This section describes in general terms the routes of exposure of ecological receptors to chemicals on the south impoundment portion of the Site. This information provides the basis for the CSM for ecological exposures in the area of the south impoundment.

2.4.1 Ingestion

Direct ingestion of chemicals is commonly used to evaluate exposure in an ERA because much of the available and relevant toxicity literature for birds and mammals reports on the oral toxicity of chemicals and because many receptors ingest multiple contaminated media (i.e., food, water, and soil). The oral dose is considered greatest among the possible exposure routes for most terrestrial species.

Reptiles ingest soil directly while burrowing or foraging. Birds and mammals can ingest soil directly while foraging and cleaning their fur or feathers (Beyer et al. 1994). Reptiles, birds, and mammals also ingest bioaccumulative COIs through consumption of contaminated prey tissue. The extent to which trophic transfer via ingestion occurs is dependent on numerous factors, including the exposure of the prey to COIs, the bioaccumulation potential of the specific chemical, the extent to which the chemical is partitioned in the tissues of the prey, and what parts of the prey are eaten by the receptor. Trophic transfer is of particular importance for hydrophobic bioaccumulative chemicals of concern and for higher trophic-level consumers (e.g., raptors and carnivorous mammals).

2.4.2 Direct Contact

For terrestrial ecological receptors, direct contact exposure may include uptake across the integument (an enveloping layer such as a skin, membrane, or cuticle). The extent of direct contact with the exposure medium depends on the chemical and the physiology, habitat, and life history characteristics of the ecological receptor. Although direct contact exposure via transfer across external tissues is possible in ecological receptors, it is rarely quantified directly in ERAs because data are not available in the literature to interpret the toxicity resulting from direct contact for most chemicals. Instead, more general means of evaluating exposure-response relationships are used. For example, in a bioassay in which the exposure is to soil, a test organism may be exposed via dermal uptake, and ingestion of the contaminated soil. However, only the concentration in the soil is measured and this concentration is used to evaluate the threshold "exposure" associated with effects. Exposures via each route are never quantified or reported, and may not be needed to interpret the results. Due to a fundamental lack of information to differentiate direct contact exposures from other routes in exposures of ecological receptors, and to interpret this specific exposure route, absorption across the integument is not explicitly addressed by this SLERA.

2.4.3 Inhalation

Inhalation is a potentially complete pathway for wildlife by inhaling airborne particulates or volatilized chemicals. Volatile chemicals are not expected to be present in surface soil in meaningful concentrations for risk, so inhalation of vapors in outdoor air is not a complete pathway. Inhalation is generally considered a relatively minor exposure pathway for wildlife relative to ingestion via soils. An evaluation of risk to receptors via the inhalation pathway may be warranted, however, in cases where volatile organic compounds are COIs and pathways of exposure are complete, including the potential for volatilization of chemicals and exposure to burrowing animals in subsurface soils.

2.5 Assessment Endpoints

An assessment endpoint is "an explicit expression of the environmental value to be protected, operationally defined as an ecological entity and its attributes" (USEPA 2003). Clearly defined assessment endpoints help structure an ERA to address management decisions.

Clarity in assessment endpoints is essential to their role in refining the direction of the risk assessment, and in communicating the meaning of the results generated by the SLERA.

USEPA guidance stipulates that assessment endpoints for a SLERA reflect a conservative evaluation of risk, and address any adverse effect potentially resulting from complete exposure pathways linking contaminants to receptors (USEPA 1998). Consistent with USEPA guidance for the SLERA, assessment endpoints are the populations of chosen receptors as inferred from measures related to survival, growth, and reproduction of individuals (USEPA 1998). A summary of assessment endpoints is presented in Table E-3.

The SLERA does not specify the extent or severity of effects of exposure to chemicals on the assessment endpoints for each receptor. Instead, the SLERA identifies those chemicals that have no potential effect on ecological receptors. By using a conservative evaluation of exposure and toxicity, the SLERA identifies those chemicals that require additional evaluation in the BERA, when more realistic and site-specific exposure and toxicity information is considered.

2.6 Preliminary Conceptual Site Model

A CSM is a summary of the sources of contaminants, the physical-chemical processes that control chemical transport and fate over time and space. The PSCR (Integral and Anchor QEA 2012) describes the most current CSM for the south impoundment (Figure E-3) and supporting rationale. It also presents the exposure pathways that potentially lead to exposures of each general category of ecological receptors to COIs. For ecological receptors using the area south of I-10, contact with, inhalation of, and ingestion of contaminated soil within the boundary of the impoundment itself, and in other areas to which COIs may have been transported, creates the potential for exposure (Figure E-4).

3 SCREENING LEVEL EVALUATION AND IDENTIFICATION OF COPC$_E$S

According to USEPA (1997) guidance for conducting SLERAs, only one of the following three conclusions can result from a SLERA for each COPC$_E$:

- There is sufficient information to conclude that ecological risks are negligible
- There is not enough information to make a decision, and additional study may be warranted
- There is adequate information to indicate that a potential for adverse effects exists, and a more thorough assessment is warranted.

A SLERA necessarily applies conservative judgments where there are data gaps or other uncertainties. A conservative approach is used so that a conclusion that ecological risks are negligible can be made with a high degree of confidence.

The screening level exposure estimate and risk calculation is Step 2 of the screening process as defined by USEPA guidance (Figure E-1). Step 2 identifies those Site-related COIs for which there is not enough information to make a decision, or which need additional assessment, and those COIs that represent negligible or no ecological risks. The methods and results used to perform the screening analysis for the data collected to date are presented below. These methods and results have already been presented in Attachment 1 to Soil SAP Addendum 3 (Integral 2011a), and directly reflect the DQOs for the Phase I soil investigation for the south impoundment (Integral 2011b). This document repeats those results, and builds on them to identify COPC$_E$s for the south impoundment.

According to the DQOs for the soil investigation south of I-10 and the analysis path described by Integral (2011b), the analytical approach for the Phase I results of the soil investigation includes the following steps:

- Evaluation of detection frequency. Chemicals detected at a frequency of 5 percent or less are not addressed by subsequent analyses.
- Risk-based screening, consisting of comparison of COI concentrations in surface and shallow subsurface soils to screening levels protective of ecological receptors.

This section presents the results of these analysis steps, and presents the rationale for selection of those COPCEs to be evaluated in the BERA. For the screening comparison, COI concentrations only in surface soils (0 to 6 inches) and shallow subsurface soils (6 to 12 inches) are compared to ecological risk-based screening levels. Risks associated with COPCEs will be evaluated in the BERA for the area south of I-10 according to receptor-specific exposure assumptions, which may include consideration of deeper soils.

3.1 Detection Frequency

COIs with a detection frequency at or below 5 percent of all samples collected in Phase I will not be considered further for the south impoundment area. Detection frequencies for each COI are reported in Table E-4. All of the metals were detected in more than 5 percent of samples, as were dioxin and furan congeners, three of the nine Aroclors, and several semivolatile organic compounds (SVOCs). Those with a detection frequency of 5 percent or less were SVOCs and the two volatile organic compounds on the COI list. The following COIs were detected in 5 percent of samples or less:

- 2,4-Dichlorophenol
- 1,2,3-Trichlorobenzene
- 1,2,4-Trichlorobenzene
- 2,4,5-Trichlorophenol
- 2,4,6-Trichlorophenol
- 2,3,4,6-Tetrachlorophenol
- Pentachlorophenol
- Hexachlorobenzene
- Chloroform.

Although several Aroclors were never detected (Aroclors 1016, 1221, 1232, 1248, 1262, and 1268), any one Aroclor represents a mixture of polychlorinated biphenyls (PCBs), and one or more PCB congeners making up an Aroclor mixture may be present, even if the Aroclor is not detected. Therefore, the detection frequency of zero for these Aroclors does not provide the basis for eliminating any individual PCB congeners from further analysis.

3.2 Ecological Risk-Based Screening Methods

The CSM indicates that terrestrial mammals, reptiles, and birds are the ecological receptor categories of interest for the south impoundment area. To perform a screening evaluation in support of identification of COPCES, soil screening values protective of birds and mammals were assembled from USEPA's ecological soil screening levels (EcoSSL) (USEPA 2005) (Table E-5). USEPA's EcoSSLs are preferred because they are the result of a rigorous and transparent process involving comprehensive literature assembly and review. Unfortunately, rigorously derived soil screening levels for PCBs and SVOCs were not found (Table E-5).

TCEQ (2006) guidance allows for the use of Texas-specific median background concentrations to screen out COIs when no screening values are available. If concentrations on the Site are below the Texas median background concentration, screening levels may be ignored. If no EcoSSLs or Texas-specific median background concentrations were available, which was the case for all of the SVOCs as well as PCBs, the median value for the Site-specific background concentrations for surface and shallow subsurface soils was used for comparison. These are shown in Tables E-5 and E-6. Consideration of the Site-specific median background concentrations is consistent with TCEQ guidance cited above (TNRCC 2001; TCEQ 2006).

Only screening levels for birds and mammals are used in this document. Soil screening levels for reptiles are generally not available, and, for the purposes of screening, the screening values for birds are considered to be protective of reptiles. In addition, rigorous and technically defensible ecological soil screening levels for dioxins and furans, or toxicity equivalents for dioxins and furans, were not found for this analysis. Therefore, dioxins and furans are not considered in this ecological screen, but are considered COPCES for the south impoundment area, and will be addressed by the BERA.

Once all of the screening or background median values were compiled, the maximum concentration among all samples from 0 to 6 and 6 to 12 inches were compared to the screening value (Table E-6). Consistent with ERA guidance (USEPA 1998), maximum concentrations or, in the case where the chemical was not detected in all Site samples, one-half the maximum detection limit was used to provide a conservative estimate of exposure concentrations for ecological receptors for the screening evaluation.

3.3 Ecological Risk-Based Screening Results

Results of comparisons of COI concentrations in soils from 0- to 6- and 6- to 12-inch intervals to ecological soil screening values are summarized in Table E-6:

- The following COIs were not found at concentrations greater than the screening value for mammals: aluminum, arsenic, barium, cobalt, manganese, nickel, silver, and vanadium.
- The following COIs were not found at concentrations above the screening value for birds: aluminum, antimony, arsenic, cobalt, manganese, nickel, and silver.

Several metals are present in one or more soil sample from 0 to 6 or 6 to 12 inches at concentrations greater than screening values:

- The following are present at least once at a concentration greater than the screening level for mammals: antimony, cadmium, chromium, copper, mercury, lead, thallium, and zinc.
- The following are present above screening concentrations for birds, in one or more samples: barium, cadmium, chromium, copper, lead, mercury, thallium, vanadium, and zinc.

Screening for antimony, barium, chromium, mercury, and thallium was performed using the Site-specific median background concentrations, because neither ecological risk-based screening values nor Texas median background concentrations were available (Table E-6). From these comparisons, it is evident that magnesium, total PCBs, and all of the SVOCs are present at concentrations greater than the median of the Site-specific background dataset. The exceedances of the median concentration by 1,2-dichlorobenzene, 1,3-dichlorobenzene, and 1,4-dichlorobenzene are not considered likely to be ecologically significant because the maxima for these analytes are nondetects, and the differences from the background medians are very slight (Table E-6).

3.4 Identification of $COPC_{E}$s for the South Impoundment

To conduct the BERA, it is necessary to identify $COPC_{ES}$. For the south impoundment, the approach to determining whether each COI is a $COPC_{E}$ is similar to the approach described in Appendix C of the RI/FS Work Plan, and consistent with that approach:

- If the maximum concentration of a COI is greater than the soil screening level or greater than the median background concentration (for those COIs lacking screening levels), the chemical is considered a $COPC_{E}$ if it is bioaccumulative.
- If a COI is not bioaccumulative, then it will not be evaluated in the BERA, even if it exceeds a soil screening level or median background concentration.

All resulting $COPC_{ES}$ will be included in the BERA for the south impoundment.

Potential for bioaccumulation of metals was evaluated using TCEQ guidance, which lists chemicals considered to be bioaccumulative (Table 3-1 in TNRCC [2001] and TCEQ [2006]). Because TCEQ guidance does not address some of the organic COIs, for all of the organic COIs, the log K_{ow} was used as an indicator of bioaccumulation potential. Consistent with USEPA (2008a) guidance, chemicals with log K_{ow}s equal to or greater than 5 were considered to have the potential to bioaccumulate in tissue.

A summary of the decision for each chemical is provided in Table E-7. The following chemicals exceeded a screening concentration or background, but were not selected as $COPC_{ES}$ because they are not considered to be bioaccumulative:

- Antimony. This metal was present at a concentration above the EcoSSL for mammals, but it does not exceed the Texas median background concentration and it is not bioaccumulative, and so it was not considered a $COPC_{E}$.
- Barium. This metal was present below the EcoSSL for mammals and slightly above the EcoSSL for birds, but is not bioaccumulative, and so was not considered a $COPC_{E}$. The maximum concentration is below the Site-specific reference envelope value (Integral and Anchor QEA 2012).
- Magnesium. This metal was present above the Site-specific background concentration, but it did not exceed the reference envelope value or the mean in the

Site-specific background soils data. It is not bioaccumulative and is an essential nutrient for many species. It was not considered a COPCE.

- Thallium. This metal was present at concentrations greater than the Texas median background concentration, and greater than Site-specific background, but is not considered bioaccumulative, and so was not considered a COPCE.

A summary of the final COPCES for the south impoundment is presented in Table E-8.

Because USEPA has requested that additional information be collected to describe COIs in south impoundment soils (Integral 2011a), results of any future sample collection and analyses will be considered before a final determination of COPCES for the south impoundment is made.

4 SCIENTIFIC MANAGEMENT DECISION POINT

According to USEPA (1997) guidance, the end of Step 2 of the ERA process is an SMDP, and a decision is made about those chemicals for which more information may be needed, and those chemicals for which there is enough information to make a determination of negligible risk. This SLERA concludes that there is not enough information to make a determination about ecological risk for the following chemicals:

- Dioxins and furans
- PCBs
- Bis(2-ethylhexyl)phthalate
- Cadmium
- Chromium
- Copper
- Lead
- Mercury
- Zinc.

For all other chemicals, the available information indicates that ecological risks are negligible.

In this SLERA, uncertainties were mitigated by the use of the following specific conservative approaches, methods, or assumptions:

- Development of a comprehensive COI list for the starting point for screening, as described in Appendix C to the RI/FS Work Plan (Anchor QEA and Integral 2010). On the basis of a list of the priority pollutants possibly in the source material at the Site, a conservative set of criteria was used to identify and define the COIs before the risk-based screening process was applied.
- Use of chemistry information for soils collected at the most likely location of the former south impoundment, which could reasonably be expected to have the highest concentrations of COIs in surface soils, to represent the screening level exposures.
- Use of the maximum concentration of each chemical in soil from within the south impoundment perimeter and within areas accessible to wildlife (surface soils) in the screening.

Results of the Phase I and any subsequent soil investigations will be incorporated into the BERA for the south impoundment to reduce uncertainties and establish a more realistic assessment of ecological risks.

5 REFERENCES

Anchor QEA and Integral, 2010. Final Remedial Investigation/Feasibility Study Work Plan, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper, and U.S. Environmental Protection Agency, Region 6. Anchor QEA, Ocean Springs, MS, and Integral Consulting Inc., Seattle, WA. November 2010.

ATSDR, 1998. Toxicological Profile for Chlorinated Dibenzo-*p*-Dioxins. Draft Report. U.S. Department of Health and Human Services, Public Health Service, Agency for Toxic Substances and Disease Registry. December 1998.

Beyer, W.N., E.E. Connor, and S. Gerould, 1994. Estimates of soil ingestion by wildlife. *Journal of Wildlife Management* 58:375-382.

Cannatella, D., and T. LaDuc. 2011. Herps of Texas, Texas Natural Science Center, University of Texas, Austin. On-line publication accessed at: http://www.herpsoftexas.org/content/common-gartersnake on October, 24, 2011.

Croteau, M., S.N. Luoma, and A.R Stewart, 2005. Trophic transfer of metals along freshwater food webs: Evidence of cadmium biomagnification in nature. *Limnol. Oceanogr.* 50(5):1511-1519.

Davis, W.B., and D.J. Schmidly, 1994. The Mammals of Texas: Online Edition. Accessed at http://www.nsrl.ttu.edu/tmot1/Default.htm. Accessed on September 27, 2011.

Fan et al., 2006. Fate and transport of 1278-TCDD, 1378-TCDD, and 1478-TCDD in soil-water systems. *Science of the Total Environment* 371:323-333.

Georgia DNR, 2012. Georgia Department of Natural Resources, Wildlife Services Division. Accessed at http://www.georgiawildlife.com/node/937 on July 30, 2012

Integral, 2010. Technical Memorandum on Bioaccumulation Modeling, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September 2010.

Integral, 2011a. Sampling and Analysis Plan: Soil Study – Addendum 3, San Jacinto River Waste Pits Superfund Site. Prepared for International Paper Company and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. February 2011.

Integral, 2011b. Sampling and Analysis Plan: Soil Study – Addendum 1, San Jacinto River Waste Pits Superfund Site. Prepared for International Paper Company and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. February 2011.

Integral and Anchor QEA, 2012. Preliminary Site Characterization Report, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA, and Anchor QEA, LLC, Ocean Springs, MS. February 2012.

Jackson, B.J., and J.A. Jackson, 2000. Killdeer (*Charadrius vociferus*), The Birds of North America Online. A. Poole (ed.). Ithaca: Cornell Lab of Ornithology; Retrieved from the Birds of North America Online: http://bna.birds.cornell.edu/bna/species/517.

KSR, 2011. Mammals of Kansas, Plains pocket gopher (*Geomys bursarius*). Online publication accessed at: http://www.ksr.ku.edu/libres/mammals_of_kansas/geomys-burs.html. Accessed on October 11, 2011.

McAtee, W.L., and F.E.L. Beal, 1924. Some common game, aquatic, and rapacious birds in relation to man. *U.S.D.A. Farmers' Bull.* No. 497 (revised).

MNH, 2012. North American Mammals, Online publication. R. Costello and A. Rosenberger, Eds. Smithsonian National Museum of Natural History. Accessed at: http://www.mnh.si.edu/mna/image_info.cfm?species_id=97 on October 19, 2011.

Suedel, B.C., J.A. Boraczek, R.K. Peddicord, P.A. Clifford, and T.M. Dillon, 1994. Trophic transfer and biomagnification potential of contaminants in aquatic ecosystems. *Reviews of Environmental Contamination and Toxicology* 136:21–89.

Sulentich, J.M., L.R. Williams, and G.N. Cameron, 1991. Mammalian Species No. 383, *Geomys breviceps*. The American Society of Mammalogists, November 6. Available at: http://www.science.smith.edu/departments/Biology/VHAYSSEN/msi/pdf/i0076-3519-383-01-0001.pdf.

TCEQ, 2006. Update to Guidance for Conducting Ecological Risk Assessments at Remediation Sites in Texas. RG-263 (Revised). Texas Commission on Environmental Quality, Remediation Division. January. http://www.tceq.state.tx.us/assets/public/remediation/eco/0106eragupdate.pdf

TNRCC, 2001. Guidance for Conducting Ecological Risk Assessments at Remediation Sites in Texas. Texas Natural Resource Conservation Commission, Toxicology and Risk Assessment Section.

TPWD, 2012. Virginia Opossum (*Didelphis virginiana*). Texas Parks and Wildlife. Available at: http://www.tpwd.state.tx.us/huntwild/wild/species/oposum/ Accessed on July 30, 2012.

TSDH, 1966. Investigation of Industrial Waste Disposal – Champion Paper, Inc. Pasadena. Texas State Department of Health Memorandum from Stanley W. Thompson, P.E., Regional Engineer, to the Director of the Division of Water Pollution Control. May 6, 1966.

TSHA, 2009. The San Jacinto River. Texas State Historical Association. Accessed at: http://www.tshaonline.org/handbook/online/articles/SS/rns9.html. Accessed on December 25, 2009.

UMMZ, 2011. Diversity Web. http://animaldiversity.ummz.umich.edu. Accessed September 26, 2011. University of Michigan, Museum of Zoology, Ann Arbor, MI.

USAF, 2006. Remedial Investigation Report, Bellows Hardfill (LF23) Bellows Air Force Station, Oahu, Hawaii. April 2006.

USEPA, 1997. Ecological Risk Assessment Guidance for Superfund, Process for Designing and Conducting Ecological Risk Assessments. USEPA/540-R-97-006. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1998. Guidelines for Ecological Risk Assessment. EPA/630/R-95/002F. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2000. Exposure and Human Health Reassessment of 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (TCDD) and Related Compounds. EPA/600/P-00/001Bc. Draft Final Report. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Washington, DC.

USEPA, 2003. Generic Ecological Assessment Endpoints for Ecological Risk Assessment. EPA/630/P-02/004F. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2005. Guidance for Developing Ecological Soil Screening Levels. OSWER Directive 9285.7-55. U.S. Environmental Protection Agency, Office of Solid Waste and Emergency Response.

USEPA, 2007. Framework for Metals Risk Assessment. EPA 120/R-07/001. U.S. Environmental Protection Agency, Washington, DC.

USEPA, 2008a. Using Fish Tissue Data to Monitor Remedy Effectiveness. OSWER Directive 9200.1-77D. Sediment Assessment and Monitoring Sheet (SAMS) #1. U.S. Environmental Protection Agency, Office of Superfund Remediation and Technology Innovation and Office of Research and Development, Washington, DC.

USEPA, 2008b. Framework for Application of the Toxicity Equivalence Methodology for Polychlorinated Dioxins, Furans, and Biphenyls in Ecological Risk Assessment. EPA 10/R-0 /00-200. U.S. Environmental Protection Agency.

USEPA, 2009. Unilateral Administrative Order for Remedial Investigation/Feasibility Study. U.S. EPA Region 6 CERCLA Docket No. 06-03-10. In the matter of: San Jacinto River Waste Pits Superfund Site Pasadena, Texas. International Paper Company, Inc., and McGinnes Industrial Management Corporation, respondents.

USEPA, 2011. Regional Screening Levels for Industrial Soil. U.S. Environmental Protection Agency. Available at: http://www.epa.gov/reg3hwmd/risk/human/rb-concentration_table/Generic_Tables/index.htm. Accessed July 20, 2011.

W&M, 2011. City of Houston Municipal Setting Designation Application. Prepared for Southwest Shipyard, L.P., Channelview, Texas. W&M Environmental Group, Inc., Houston, Texas.

Zimmerman, R., 2002. *Thamnophis sirtalis* (On-line), Animal Diversity Web. Accessed at: http://animaldiversity.ummz.umich.edu/site/accounts/information/Thamnophis_sirtalis.html. Accessed on October 24, 2011.

TABLES

Table E-1
Chemicals of Interest

Class	Chemical
Metals	
	Aluminum
	Antimony
	Arsenic
	Barium
	Cadmium
	Chromium
	Cobalt
	Copper
	Lead
	Magnesium
	Manganese
	Mercury
	Nickel
	Silver
	Thallium
	Vanadium
	Zinc
Dioxins and Furans	
	Dioxins and Furans
Polychlorinated Biphenyls	
	Polychlorinated Biphenyls
Semivolatile Organic Compounds	
	2,4-Dichlorophenol
	2,4,5-Trichlorophenol
	2,4,6-Trichlorophenol
	2,3,4,6-Tetrachlorophenol
	Acenaphthene
	Bis(2-ethylhexyl)phthalate
	Carbazole
	Fluorene
	Hexachlorobenzene
	Naphthalene
	Pentachlorophenol
	Phenanthrene
	Phenol
Volatile Organic Compounds	
	1,2-Dichlorobenzene
	1,3-Dichlorobenzene
	1,4-Dichlorobenzene
	1,2,3-Trichlorobenzene
	1,2,4-Trichlorobenzene
	Chloroform

Table E-2
Ecological Receptor Surrogates for the South Impoundment

Receptor Group	Receptor Surrogate	Feeding Guild	Potentially Present	Representative of One or More Feeding Guilds	High Site Fidelity/Residential	Sensitive or Potentially Highly Exposed	Life History Information Is Readily Available	Additional Considerations
Reptiles								
	Common garter snake	Omnivore (terrestrial)	X	X	X	X	X	
Birds								
	Killdeer	Invertivore (terrestrial)	X	X	X	X	X	Feeds on invertebrate fauna closely associated with soils
Mammals								
	Virginia opossum	Omnivore (terrestrial)	X	X	X	X	X	Terrestrial omnivore with habitat preferences consistent with site conditions
	Pocket gopher	Herbivore	X	X	X	X	X	Burrowing mammal, susceptible to exposure to chemicals of interest in soil

Table E-3
Summary of Receptor Surrogates, Assessment Endpoints, and Risk Questions

Receptor Class	Assessment Endpoint	Risk Questions
Reptiles	Stable or increasing populations of omnivorous reptiles	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth and reproduction of reptiles?
Birds	Stable or increasing populations of terrestrial invertivorous birds	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth, and reproduction of birds? Is the estimated concentration of dioxins and furans, expressed as TEQs, in bird eggs greater than threshold concentrations for reproductive effects in birds?
Mammals	Stable or increasing populations of omnivorous mammals	Is the total daily ingested dose (mg/kg bw-day) of COPCs greater than doses known to cause effects on the survival, growth and reproduction of mammals?
	Stable or increasing populations of burrowing mammals	Is exposure via inhalation of volatile COPCs greater than doses known to cause effects on the survival, growth and reproduction of mammals?

Notes
COPC = chemical of potential concern
TEQ = toxic equivalent

Table E-4
Detection Frequencies for Chemicals of Interest

Chemical of Interest	Detection Frequency	
Metals		
Aluminum	72/72	100%
Antimony	58/72	81%
Arsenic	72/72	100%
Barium	72/72	100%
Cadmium	69/72	96%
Chromium	72/72	100%
Cobalt	72/72	100%
Copper	72/72	100%
Lead	72/72	100%
Magnesium	71/71	100%
Manganese	72/72	100%
Mercury	71/72	99%
Nickel	72/72	100%
Silver	10/72	14%
Thallium	42/72	58%
Vanadium	72/72	100%
Zinc	72/72	100%
Dioxins and Furans		
TEQ_{DF}	94/94	100%
Polychlorinated Biphenyls		
Aroclor 1016	0/72	**0%**
Aroclor 1221	0/72	**0%**
Aroclor 1232	0/72	**0%**
Aroclor 1242	7/72	10%
Aroclor 1248	0/72	**0%**
Aroclor 1254	22/72	31%
Aroclor 1260	28/72	39%
Aroclor 1262	0/72	**0%**
Aroclor 1268	0/72	**0%**
Semivolatile Organic Compounds		
2,4-Dichlorophenol	0/72	**0%**
2,4,5-Trichlorophenol	0/72	**0%**
2,4,6-Trichlorophenol	0/72	**0%**
2,3,4,6-Tetrachlorophenol	0/72	**0%**
Acenaphthene	50/70	71%
Bis(2-ethylhexyl)phthalate	51/70	73%
Carbazole	37/70	53%
Fluorene	50/70	71%
Hexachlorobenzene	3/70	**4%**
Naphthalene	36/70	51%
Pentachlorophenol	0/72	**0%**

Table E-4

Detection Frequencies for Chemicals of Interest

Chemical of Interest	Detection Frequency	
Phenanthrene	65/70	93%
Phenol	13/72	18%
Volatile Organic Compounds		
1,2-Dichlorobenzene	14/72	19%
1,3-Dichlorobenzene	15/72	21%
1,4-Dichlorobenzene	13/72	18%
1,2,3-Trichlorobenzene	0/72	**0%**
1,2,4-Trichlorobenzene	2/72	**3%**
Chloroform	2/72	**3%**

Notes

TEQ_{DF} = Toxicity equivalent for dioxins and furans calculated using mammalian toxicity equivalency factors

Bold typeface indicates that the frequency of detection is less than or equal to 5 percent

Table E-5
Soil Screening Levels Used for Ecological Screening

Analyte	Mammalian Eco-SSL[a]	Avian Eco-SSL[a]	Background Soil Concentration[b]
Metals (mg/kg - dw)			
Aluminum	NA	NA	**30,000**
Antimony	**0.27**	NA	**1**
Arsenic	**46**	**43**	5.9
Barium	**2,000**	NA	**300**
Cadmium	**0.36**	**0.77**	NA
Chromium (total)	NA	NA	**30**
Cobalt	**230**	**120**	7
Copper	**49**	**28**	15
Lead	**56**	**11**	15
Magnesium	NA	NA	NA
Manganese	**4,000**	**4,300**	300
Mercury	NA	NA	**0.04**
Nickel	**130**	**210**	10
Silver	**14**	**4.2**	NA
Thallium	NA	NA	**0.7**
Vanadium	**280**	**7.8**	50
Zinc	**79**	**46**	30
Polychlorinated Biphenyls (µg/kg-dw)			
Aroclor 1242	NA	NA	NA
Aroclor 1254	NA	NA	NA
Aroclor 1260	NA	NA	NA
Total PCBs	NA	NA	**9.5**[d]
Semivolatile Organic Compounds (µg/kg-dw)			
1,2-Dichlorobenzene	NA	NA	**0.048**[c]
1,3-Dichlorobenzene	NA	NA	**0.06**[c]
1,4-Dichlorobenzene	NA	NA	**0.055**[c]
Acenaphthene	NA	NA	**0.7**
Bis(2-ethylhexyl)phthalate	NA	NA	**5.35**
Carbazole	NA	NA	**0.65**
Fluorene	NA	NA	**0.55**
Naphthalene	NA	NA	**1.15**
Phenanthrene	NA	NA	**2.4**
Phenol	NA	NA	**1.4**[e]

Notes

Bold = value was used in the screening evaluation (Table E-6)

a - Value is the minimum value available from the two of the feeding guilds within the taxon evaluated by USEPA (2011).

b - Values for metals are from Texas-Specific Median Background Concentration (Figure 30 TAC §350.51(m)); values for organics are site-specific median background concentration

c - Analyte was never detected in 0- to 12-inch background soils; value shown is the median of the estimated values (i.e., one-half of detection limit) for the chemical in background samples from 0 to 6 inches.

d - This value is the detection limit for individual Aroclors.

e - Detected in 1 of 40 samples.

Table E-6
Ecological Screening Results for Surface and Shallow Subsurface Soils

Chemical of Interest	Maximum Detected Concentration, Surface and Shallow Subsurface Soils (0 to 6 and 6 to 12 inch)	Ecological Screening Value, Mammals [a]	Maximum Exceeds Screening Value	Ecological Screening Value, Birds [a]	Maximum Exceeds Screening Value	Median for Background Soils (0 to 12 inch)	Maximum Exceeds Site-Specific Median Background
Metals (mg/kg - dw)							
Aluminum	11,700	30,000[b]		30,000[b]			
Antimony	1.00 J	0.27	X	1[b]			
Arsenic	5.28 J	46		43			
Barium	413 J	2,000		300[b]	X		
Cadmium	1.28	0.36	X	0.77	X		
Chromium	70.3 J	30[b]	X	30[b]	X		
Cobalt	22.1	230		120			
Copper	121 J	49	X	28	X		
Lead	117 J	56	X	11	X		
Magnesium	9,150	NA	--	NA	--	942	X
Manganese	2,630 J	4,000		4,300			
Mercury	0.156	0.04[b]	X	0.04[b]	X		
Nickel	85.1	130		210			
Silver	0.800 J	14		4.2			
Thallium	9.80 J	0.7[b]	X	0.7[b]	X		
Vanadium	52.1	280		7.8	X		
Zinc	4,160 J	79	X	46	X		
Polychlorinated Biphenyls (µg/kg-dw)							
Total PCBs	427	NA	--	NA	--	9.5	X
Semivolatile Organic Compounds (µg/kg-dw)							
1,2-Dichlorobenzene	0.055 U	NA	--	NA	--	0.048[c]	X
1,3-Dichlorobenzene	0.07 U	NA	--	NA	--	0.06[c]	X
1,4-Dichlorobenzene	0.06 U	NA	--	NA	--	0.055[c]	X
Acenaphthene	88	NA	--	NA	--	0.7	X
Bis(2-ethylhexyl)phthalate	2,200	NA	--	NA	--	5.35	X
Carbazole	48	NA	--	NA	--	0.65	X
Fluorene	46	NA	--	NA	--	0.55	X
Naphthalene	50	NA	--	NA	--	1.15	X
Phenanthrene	450	NA	--	NA	--	2.4	X
Phenol	6.5 U	NA	--	NA	--	1.4[d]	X

Notes

-- = uncertain; no screening value is available for this chemical

NA = no screening value available

U = not detected

X = maximum concentration exceeds screening value

a - USEPA's (2005) EcoSSLs were used, and where they were not available, Texas Median Background concentration is shown (Table E-5)

b - The Texas median background concentration is shown.

c - Analyte was never detected in 0- to 12-inch background soils; value shown is the median of the estimated values (i.e., one-half of detection limit) for the chemical in background samples from 0 to 6 inches.

d - Detected in 1 of 40 samples.

Table E-7
Selection of $COPC_E$s for the South Impoundment

Chemical	Log Kow of Chemical (Organics Only) [a]	Is Chemical Potentially Bioaccumulative? [b]	Maximum Exceeds Avian Screening Value or Background	Maximum Exceeds Mammalian Screening Value or Background	Maintain as COPC for South Impoundment Receptors?		Reason for COPC Decision
					Birds and Reptiles	Mammals	
Metals (mg/kg)							
Aluminum	NA	No	No	No	No	No	Not potentially bioaccumulative
Antimony	NA	No	No	Yes	No	No	Not potentially bioaccumulative
Arsenic	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Barium	NA	No	B	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSL for mammals
Cadmium	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Chromium	NA	Yes	B	B	Yes	Yes	Potentially bioaccumulative, exceeds Texas Median Background
Cobalt	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Copper	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Lead	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Magnesium	NA	No	B	B	No	No	Not potentially bioaccumulative
Manganese	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Mercury	NA	Yes	B	B	Yes	Yes	Potentially bioaccumulative, exceeds Texas Median Background
Nickel	NA	Yes	No	No	No	No	Potentially bioaccumulative, but did not exceed mammal or bird EcoSSLs
Silver	NA	No	No	No	No	No	Not potentially bioaccumulative, did not exceed EcoSSLs
Thallium	NA	No	B	B	No	No	Not potentially bioaccumulative
Vanadium	NA	No	Yes	No	No	No	Not potentially bioaccumulative, did not exceed mammal EcoSSL
Zinc	NA	Yes	Yes	Yes	Yes	Yes	Potentially bioaccumulative, exceeds bird and mammal EcoSSLs
Dioxins/Furans (ng/kg)	>5	Yes	NA	NA	Yes	Yes	Potentially bioaccumulative, indicator chemical group
Polychlorinated Biphenyls (µg/kg)	>5	Yes	B	B	Yes	Yes	Potentially bioaccumulative, detected above background
Semivolatile Organic Compounds (µg/kg)							
2,4-Dichlorophenol	3.06	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,4,5-Trichlorophenol	3.69	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,4,6-Trichlorophenol	3.72	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
2,3,4,6-Tetrachlorophenol	4.45	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
Acenaphthene	3.92	No [c]	B	B	No	No	Not potentially bioaccumulative
Bis(2-ethylhexyl)phthalate	7.6	Yes	B	B	Yes	Yes	Potentially bioaccumulative, present above the Site-specific background median concentration
Carbazole	3.72	No [c]	B	B	No	No	Not potentially bioaccumulative
Fluorene	4.18	No [c]	B	B	No	No	Not potentially bioaccumulative
Hexachlorobenzene	5.73	Yes	NA	NA	No	No	Detected in less than 5% of samples
Naphthalene	3.3	No [c]	B	B	No	No	Not potentially bioaccumulative
Pentachlorophenol	5.12	Yes	NA	NA	No	No	Detected in less than 5% of samples
Phenanthrene	4.57	No [c]	B	B	No	No	Not potentially bioaccumulative
Phenol	1.46	No	B	B	No	No	Not potentially bioaccumulative

Table E-7
Selection of $COPC_E$s for the South Impoundment

Chemical	Log Kow of Chemical (Organics Only) [a]	Is Chemical Potentially Bioaccumulative? [b]	Maximum Exceeds Avian Screening Value or Background	Maximum Exceeds Mammalian Screening Value or Background	Maintain as COPC for South Impoundment Receptors?		Reason for COPC Decision
					Birds and Reptiles	Mammals	
Volatile Organic Compounds (µg/kg)							
1,2-Dichlorobenzene	3.43	No [c]	B	B	No	No	Not potentially bioaccumulative, maximum concentration was non-detect
1,3-Dichlorobenzene	3.53	No [c]	B	B	No	No	Not potentially bioaccumulative, maximum concentration was non-detect
1,4-Dichlorobenzene	3.44	No [c]	B	B	No	No	Not potentially bioaccumulative, max.concentration was non-detect
1,2,3-Trichlorobenzene	4.05	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
1,2,4-Trichlorobenzene	4.02	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative
Chloroform	1.97	No [c]	NA	NA	No	No	Detected in less than 5% of samples, not potentially bioaccumulative

Notes

B = Maximum concentration exceeds Texas median background concentration or Site-specific median background concentration

$COPC_E$ = chemical of potential concern for south impoundment ecological receptors

NA = not applicable

TCEQ = Texas Commission on Environmental Quality

a - Log Kow: Octanol-water partition coefficient, the ratio of the concentration of a chemical in octanol and water at equilibrium and at a specified temperature. Octanol is an organic solvent that is used as a surrogate for natural organic matter (e.g., lipids). Values obtained from the HSDB (http://toxnet.nlm.nih.gov/cgi-bin/sis/htmlgen?HSDB) or Oak Ridge National Laboratory Risk Assessment Information System (http://rais.ornl.gov/cgi-bin/tox/TOX_select?select=chem)

b - Determination of potential for bioaccumulation from soil is based on TCEQ guidance (TCEQ 2006) or, if chemical is not addressed in guidance, log Kow information is used to determine bioaccumulative potential (as indicated in footnote c), with those chemicals having log Kow>5 being considered potentially bioaccumulative (USEPA 2008).

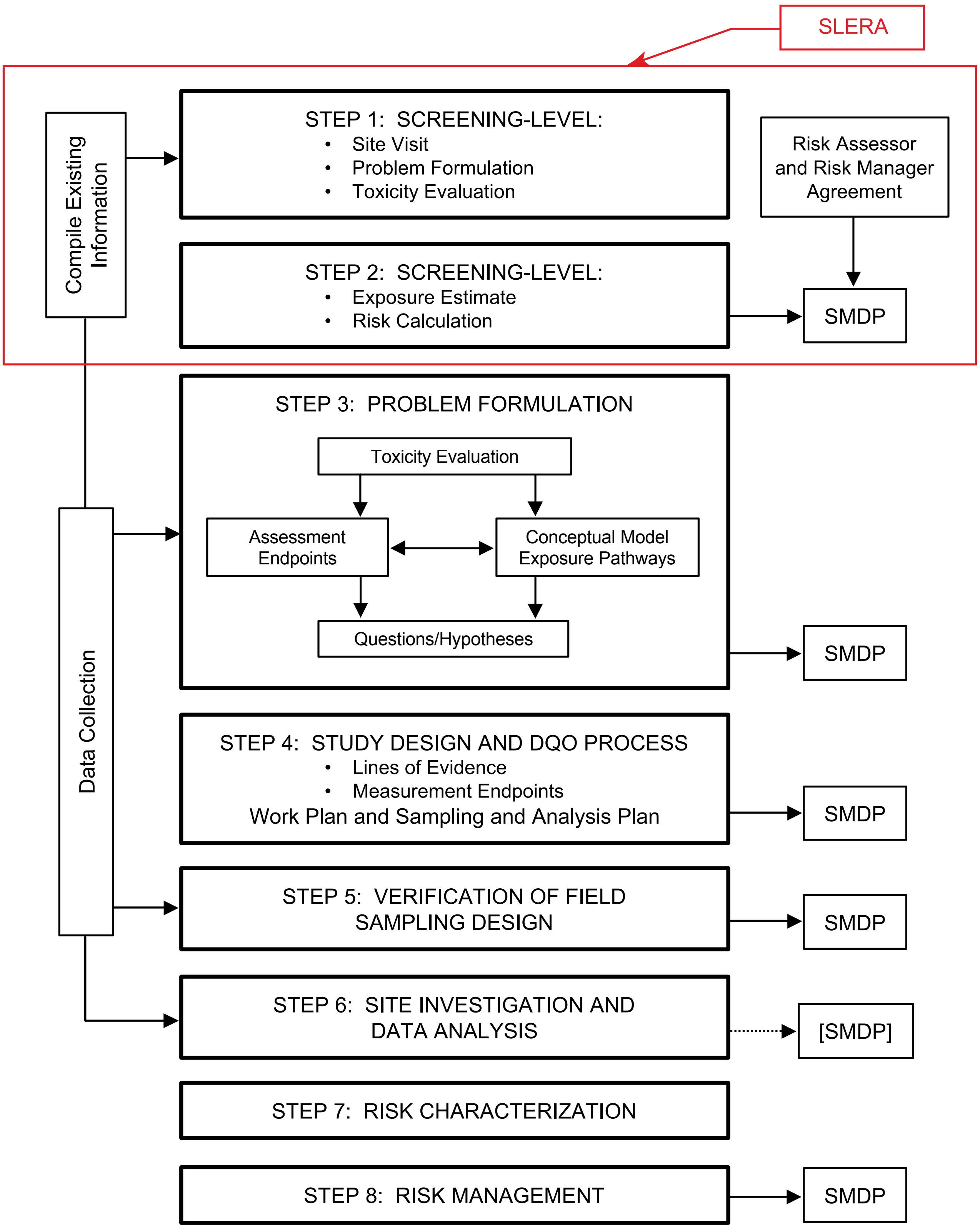
c - Not provided in TCEQ guidance; log Kow used to determine potential for bioaccumulation as described in footnote b.

Table E-8
$COPC_E$s for the South Impoundment

Chemical	Reptiles, Birds, and Mammals
Metals	
Cadmium	X
Chromium	X
Copper	X
Lead	X
Mercury	X
Zinc	X
Dioxins/Furans	
Dioxins and Furans	X
Polychlorinated Biphenyls	
Polychlorinated Biphenyls	X
Semivolatile Organic Compounds	
Bis(2-ethylhexyl)phthalate	X

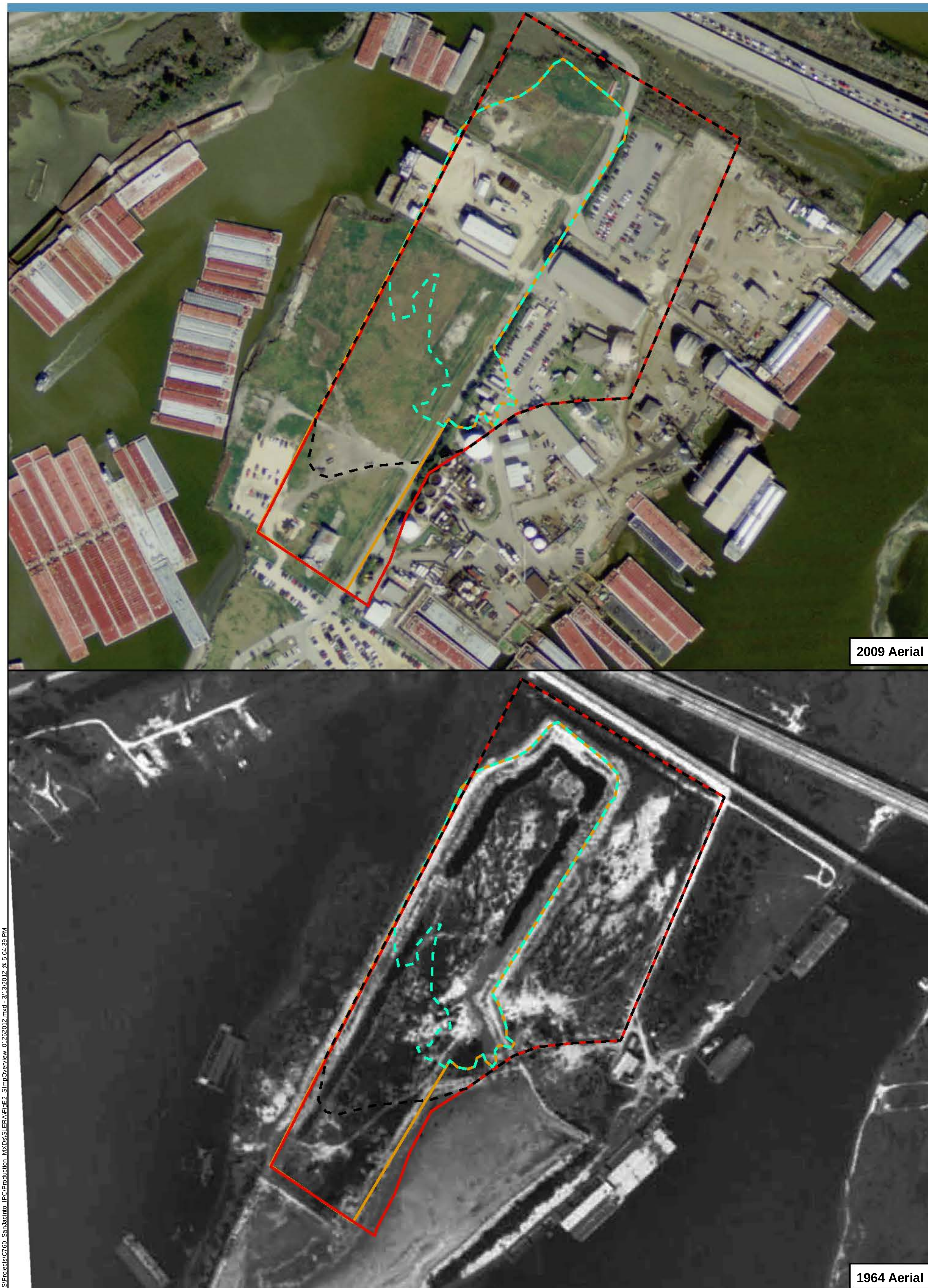
Notes
$COPC_E$ = chemical of potential ecological concern

FIGURES



[SMDP] only if change to the sampling and analysis plan is necessary.

Figure E-1
The Screening Level Ecological Risk Assessment in the Context of the USEPA 8-Step Process for Ecological Risk Assessment
SJRWP SLERA
SJRWP Superfund/IPC



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Digitized State Department of Health Hand Drawn Map

Flooded Area on 1966 Aerial Photograph

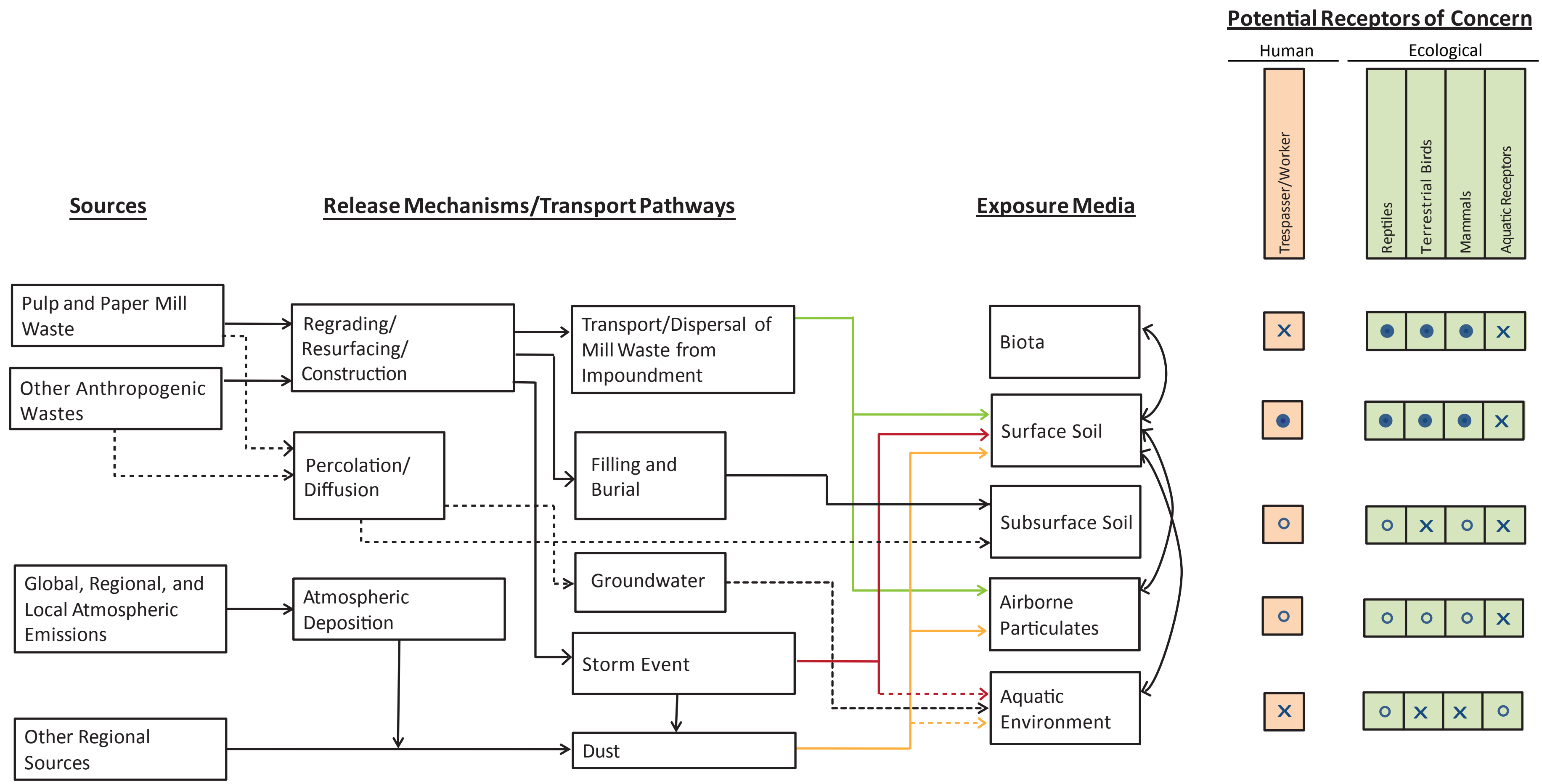
Texas State Department of Health May 1966 Hand-Drawn Map (TSDH 1966) and Soil Investigation Area 4

USGS 1964 Aerial Photograph South Impoundment Perimeter

0 400

Feet

Figure E-2
Soil Investigation Area 4
SJRWP SLERA
SJRWP Superfund/IPC



Complete, significant exposure pathway
Complete, minor exposure pathway
X Incomplete exposure pathway
Unknown pathway

Notes:
Local sources may include industrial air emissions, vehicle or machinery fluid leaks, or other releases resulting from ongoing commercial activities on the site.
Curved lines indicate potential transport pathways for chemicals of potential concern among exposure media.

Figure E-3
Updated Conceptual Site Model Pathways for the Area South of I-10
SJRWP SLERA
SJRWP Superfund/IPC

			Terrestrial Birds[a]	
Exposure Media	Exposure Routes	Reptiles	Invertivore	Mammals[a]
Biota	Ingestion	●	●	●
Soils	Ingestion	●	●	●
	Direct Contact	○	○	○
	Inhalation	○	○	●[b]

Notes:

● Potentially complete and significant exposure pathway

○ Potentially complete but minor exposure pathway

[a] Mammals and terrestrial birds are assumed not to ingest surface water for drinking, as surface water is estuarine.

[b] Potentially complete and significant for burrowing mammals only (e.g., pocket gopher).

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Figure E-4

Conceptual Site Model for Ecological Exposures:

Exposure Details for Receptor Feeding Guilds and Habitat Associations

SJRWP SLERA

SJRWP Superfund/IPC

APPENDIX F
EPA COMMENTS RELATING TO THE DRAFT BASELINE ECOLOGICAL RISK ASSESSMENT (BERA) DATED MARCH 15, 2012, AND RESPONSES, AND DRAFT-FINAL BERA DATED AUGUST 2012, AND RESPONSES

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	Section	Comment	Response to Comment—Proposed Revision
General Comments			
1		Evaluation of Threatened and Endangered Species: The EPA previously commented (see June 3, 2010 letter regarding review of the draft RI/FS Work Plan and SLERA, Comment 41) that if state or federally listed threatened or endangered wildlife species could occur in the vicinity of the Site, the SERA should designate a surrogate species for the protected species, and base any hazard quotient calculations or risk characterization on the NOAEL TRV (no-observed adverse effect level toxicity reference value) or equivalent. The PRPs agreed with the response and indicated that the text of Appendix B and Attachment Bl would be modified to address the appropriate surrogate species for any listed species that may occur at the Site. Appendix B of the RI/FS work plan generally stated (Section 2.3.2) that the risk assessment for the protected species would not employ the use of surrogates because of the potential to overestimate risk to these listed species, that realistic exposure parameters would be identified for these species, and species specific exposures would be evaluated against the appropriate TRVs in the BERA. The BERA did imply or state (Section 3.4.4) that the sandpiper would make an appropriate representative for the white-faced ibis, a State-threatened species, due to similar feeding/foraging strategies. Because the NOAEL hazard quotients for copper [central tendency (CT) = 2; reasonable maximum exposure concentration (RM) = 3] and TEQ_{DFP} (CT = 10; RM = 30) were greater than 1, the assessment shall include a more robust discussion/ analysis (TEQ_{DFP} denotes the toxicity equivalent (TEQ) concentrations calculated using dioxins and furans and dioxin-like PCBs). The text simply states that the ibis would only be an occasional visitor to the Site and its exposure potential is considered low.	Although endangered species are addressed by the draft Baseline Ecological Risk Assessment (BERA) in the Problem Formulation (Section 3.4.4) and in the Uncertainty Analysis (Section 7.1), exposures were not quantified for those endangered species that could occasionally occur on the site (brown pelican, bald eagle, and white-faced ibis). Because the risk assessment concluded that only those organisms with small home ranges in areas near the northern impoundments are potentially exposed at levels associated with risk, a quantitative evaluation of exposure for these species was not conducted, because they have large home ranges that are much greater than the area of the San Jacinto River Waste Pits site (Site). To address the risk to protected species with greater specificity, text will be added to the problem formulation section, to the exposure assessment, to Section 6 (as Section 6.7), to the discussion in Section 7, and to Appendix A. Revisions will more clearly address exposure and risk as a function of home range size relative to that of the receptor surrogates.
2		Post TCRA (Time-Critical Removal Action) –Scenarios: Hazard quotient calculations were presented for the baseline site (before placement of the TCRA), and after TCRA placement. For the post-TCRA analysis, the evaluation assumed that $COPC_E$ (chemical of potential ecological concern) concentrations in sediments within the TCRA footprint (i.e., sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of 1-10) are equal to the median concentration of the chemical in the upstream background sediment dataset or the background soil dataset. Additionally, pre-TCRA tissue concentrations were used in post-TCRA analyses. The following shall be considered: the presumption that the Site post-TCRA will continue to remain devoid of habitat assumes that the Site will be maintained to prevent this from happening. The assessment shall consider that the Site post-TCRA will develop habitat over time.	The text in Section 3.4.3 that describes the post-TCRA habitat is not intended to suggest an assumption in the BERA that the post-TCRA environment will not provide habitat. The post-TCRA exposure scenarios assume that species will use the capped area as they have under baseline conditions. The only assumption that differs is the concentrations of $COPC_E$s in sediments within the original 1966 impoundment perimeter. Text will be edited in Section 3.4.3 and added to Section 3.8.4.3 to clarify this.
3		Estimating surface water concentrations of COPCs from sediments shall be considered a major data gap and point of uncertainty, and clarified as such in the report.	In a meeting with the U.S. Environmental Protection Agency (USEPA) and other agencies on July 18, 2012, it was discussed and agreed that this comment was not intended to indicate a requirement for additional sampling, but that estimation of chemicals of potential concern (COPCs) from sediments constitutes an uncertainty that should be further discussed. Discussion will be added to Section 7 to better describe uncertainty associated with the methods used to estimate water concentrations.
4		Figures depicting tissue sample locations shall include points at which the actual samples used in the analyses were located. The reader is unable to determine the spatial relationship between individual samples as currently depicted.	Figures 5, 6, 9, and 10 in the Field Sampling Report: Tissue Study (Integral 2011) depict the actual locations of crab and large fish tissue samples. These maps will be included in Section 4.2.5 of the draft final BERA. Small fish and clams were collected on transects; current maps depict the most specific representation of collection area available.

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	**Section**	**Comment**	**Response to Comment—Proposed Revision**
5		The presentation of the results of the BERA made it difficult to independently evaluate the risk conclusions. In particular, it would have been useful for the results to be presented in tables that included the site specific data along with the TRV or baseline values used for the assessment. By presenting the data in different locations, and by presenting primarily summaries rather than the raw data and calculations used to generate the summary data, it was challenging to trace the conclusions made in the BERA. A revision to this document shall include summary tables sufficient to allow reviewers to follow the assumptions made in the BERA.	At a meeting on July 18, 2012, USEPA clarified that this comment is intended to capture the overall sense conveyed by requests in detailed comments that in some areas, the reader would benefit from additional illustration of the methods. The reviewers suggested that the response include building from Table 3-10, which is a summary of assessment endpoints, to provide more of a "road map" to how the analyses were performed. Reviewers are reminded that Table 3-11 builds from Table 3-10 to list the lines of evidence, measures of exposure and measures of effect to be used to address each assessment endpoint. To provide additional illustration of methods, clarifications will be made to tables and figures including those called for in comments 17, 18, 26, 34, 35, 54, 65, and 72. Additional information in the form of exhibits will be added to show examples of how calculations were performed for the wildlife exposure model (to address this comment) and the bird-egg exposure model (to address comment 39 and 42).
6		It is not clear what criteria were used in the selection of toxicity references used to develop the TRVs for benthic invertebrates. References should have been prioritized by endpoint, life stage of receptor, habitat of receptor, and duration of test. Some of the references may not be appropriate for derivation of the TRV for this site (particularly those based on freshwater, acute tests). The report shall provide the selection criteria for the reference studies used.	The reader is referred to Sections 4.1 and 5.3 of the text, which describe the iterative approach to identification of toxicity reference values (TRVs) for benthos, and Section 1.4.1 of Appendix B, where the specific set of considerations used in selecting a TRV for any given chemical, including for those chemicals lacking sediment benchmarks, is discussed. Bulk sediment concentrations addressing community endpoints such as sediment benchmarks were preferred. In fact, TCEQ's preferred screening values were used for those $COPC_E$s for which they were available. If sediment quality guidelines (SQGs) were not available, ambient water quality criteria (AWQC) were used and compared to estimated pore water concentrations. Only when neither SQGs nor AWQC were available were other types of toxicity data used (i.e., from USEPA's ECOTOX database). The most protective value from the available ECOTOX data was selected from those studies addressing marine invertebrates; results for freshwater species were not used. This general approach is described in Section 5.3, and Section 5.3 cites Appendix B for details. Appendix B presents details for selection of toxicity values for each $COPC_E$ for benthic invertebrates. Specific information on the origin and derivation of each value is provided in Table 5-1. The BERA did not use a general literature search for each $COPC_E$, because concentrations of chemicals in sediments or estimated in sediment pore water were generally below the screening-level values used as described above. If concentrations of COPCs are generally below these broadly protective screening values, additional evaluation of the literature is not warranted. For dioxins and furans, the approach to evaluation of toxicity to benthos was described in detail in Attachment B2 of Appendix B to the RI/FS Work Plan; that information is also cited in BERA Section 5.3 and summarized or repeated in Appendix B of the BERA.

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	Section	Comment	Response to Comment—Proposed Revision
7		The assumption that the exposure of receptors post-TCRA will be at background levels for soil and sediment for areas outside the containment area is questionable. The report shall provide justification for why the sediment outside the footprint of the cap may already be at the upstream concentrations.	The post-TCRA analysis does not assume that exposures outside the containment area will be at concentrations comparable to upstream. The post-TCRA analysis only replaces samples within the TCRA footprint with the median of upstream concentrations. Data outside the footprint is not replaced; the reader is referred to Section 3.8.1: "Using the general assumption that $COPC_E$ concentrations in sediments within the TCRA footprint are equal to the median concentration of the chemical in the upstream background sediment dataset." Please also see Section 3.8.4.3, which states "sediment or soil samples collected from within the original 1966 perimeter of the impoundments north of I-10 are eliminated from the dataset used to estimate EPCs, and replaced with the median concentration of the chemical in the upstream background sediment dataset or from the background soil dataset, as appropriate." In a meeting with USEPA on July 18, 2012, reviewers clarified that their concern was that this assumption may be wrong (i.e., that it is unknown what conditions will evolve in terms of sediment chemistry on the cap), and that this uncertainty should be more clearly stated. Text will be added to Section 3.8.4.3 to highlight that the assumption cannot be verified with existing information. Section 7 will include a related discussion.
8		Statements that surface water quality criteria (a typical ARAR), derived to be protective of human and ecological receptors "should not override site-specific values". It shall be clarified whether or not this statement implies that site-specific values are equal to or more conservative than any ARARs. If not, these statements shall be deleted considering the requirements for ARARs and that the site is located in a dynamic and complex environment, where adequate site-specific exposure and risk assessment is difficult, at best.	The language in quotations in the comment could not be found anywhere in the document or appendices. In the meeting with USEPA on July 18, 2012, reviewers clarified that this comment was not intended for the BERA, and does not need to be specifically addressed in the context of this document.
9		The report shall include the rationale for the assumptions and conclusions included in the BERA so that they are transparent and understandable, and conservatism is demonstrated.	In the meeting with USEPA on July 18, 2012, reviewers reiterated that there was a general sense that the document should detail and provide more discussion of assumptions. Reviewers are referred to Tables 3-12 and 4-2 for key exposure assumptions; Section 4 for a narrative description of all exposure assessment methods and assumptions; and Appendix B for underlying information supporting the toxicity evaluation.
10		The report shall provide/expand its description and evaluation of food chain implications in the BERA.	It's unclear whether the comment refers to issues related to bioaccumulation of chemicals, or to changes in energy transfer across the aquatic community that could result from risks to lower trophic levels. Regardless, data for the Site do not include information to support a description or further discussion of the food web. The study and description of aquatic food webs require certain specific types of data, such as analysis of fish stomach contents or stable isotopes of nitrogen in a variety of organisms from a specified area and time. Data of this type have not been collected for the remedial investigation for this Site. It is therefore not appropriate to expand on "food chain implications" in the BERA. A section will be added to the document (as Section 6.7) to address this comment. The Technical Memorandum on Bioaccumulation Modeling (Integral 2010a) provides an extensive discussion of dioxin and furan bioaccumulation in fish, aquatic invertebrates and birds based on the published literature and analysis of data for the Houston Ship Channel. Summary information will be added as Section 6.8. The reviewer is referred to Integral (2010a) for details.

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	Section	Comment	Response to Comment—Proposed Revision
Specific Comments			
11	List of Acronyms	A definition for reasonable maximum exposure (RME) shall be added to the acronym list.	Where the acronym RME is used, it will be will be changed to RM (reasonable maximum, for consistency with "CT" for central tendency.
12	2.1	Site Setting and General Conceptual Site Models: The report states that other sources of dioxins and furans are present on the site. The report shall describe these sources.	Additional information will be included in Section 2.1 about other chemical sources on the Site. However, this topic is to be addressed in greater detail in the remedial investigation report, as required by USEPA (1988).
13	3.4	Ecosystems Potentially at Risk: Protections under the Bald and Golden Eagle Protection Act are similar in nature to that of the Endangered Species Act. As such, any surrogate (for Bald Eagle) risk characterization shall be done by comparing exposure to the NOAEL, rather than the LOAEL as presented here in the text.	Please see the response to comment 1. Risk characterization for protected species will be addressed in Section 6.
14	3.3.4	Endangered and Threatened Species at the Site: The report notes that the alligator snapping turtle is on the state list. The alligator snapping turtle's life history and occurrence shall be discussed as the other listed species are in the following paragraphs.	Text will be added to Section 3.4.4 to address the life history and occurrence of the alligator snapping turtle.
15	3.8.4.1	Calculation of Hazard Quotients: Disagree with the assertion that exposures resulting in HQ_L<1 should be characterized as "negligible." Chronic exposure in the site setting to concentrations between the NOAEL and LOAEL could result in some risk. Acceptable and "negligible" risk characterizations shall be limited to those with HQ_N<1. Also, while not being quantified, risks of mixtures of COPCs shall be addressed in the uncertainty section of this document.	The following proposal was discussed with USEPA and other reviewers on July 18, 2012: The language in this section will be changed to indicate the following interpretation: • HQ_N < 1 = risk is negligible • HQ_N > 1 and HQ_L < 1 = risk is very low • HQ_L > 1 = risk is considered present, and additional evaluation is needed to address the assessment endpoint The concept of the assessment endpoint will be specifically addressed by the additional language supporting interpretation of the HQ. In general, assessment endpoints are populations or communities of organisms, while the basis for the HQ is nearly always an individual-level TRV. Therefore, it is not appropriate to conclude or imply that an HQ_N > 1 signifies risk. To better describe risk in the situation where HQ_N > 1 > HQ_L, additional context will be provided with discussion of the toxicity information that is the basis for the TRV. In addition, statements about risk for individual COPC–receptor pairs will make the distinction between the individual and the population. The revised approach will be presented in Sections 3.8 and 6.1. Language will be added to Section 7 to address uncertainties regarding the consequences of exposure to chemical mixtures.
16	3.8.4.5	Comparison of Site Risks to Background: The BERA refers to upstream background in a dynamic, tidal setting (Table 6-2, 6-7, 6-8); but no description of the samples that constitute background levels is provided. The report shall provide this description.	A more detailed description of the background data set will be provided in Section 2.2.1.
17	4.1.1	Estimated Water Concentrations (Exposure of Benthic Macro-invertebrates): It appears that in Equation 4-2, the f_{oc} used is sample-specific. The report shall confirm this. Also, as this section deals with estimation of pore water concentrations, it shall be titled as such.	Clarification will be provided as requested.

EPA Comments Relating to the Draft Baseline Ecological Risk Assessment (BERA) Dated March 15, 2012, and Responses, and Draft-Final BERA Dated August 2012, and Responses

Comment No.	**Section**	**Comment**	**Response to Comment—Proposed Revision**
18	4.1.3	Results of the Benthic Macro-invertebrate Exposure Evaluation: The BERA shall provide a table that summarizes the estimated sediment pore water concentrations (i.e., mean, maximum, and minimum number of samples) for the various $COPC_E$s evaluated in this manner for the benthic exposure pathway.	A table will be added to Section 4.1.3 to provide estimated pore water concentrations.
19	4.2.1	$COPC_E$ Concentrations in Fish Diets: The referenced citation (Meador et al. 2010) shall reflect a 2011 date.	Clarification will be provided as requested.
20	4.2.2	Estimated Concentrations of Selected $COPC_E$s in Surface Water: Table 4-3 displays the sediment SWAC (surface area-weighted average concentration) and the estimated surface water concentration for a number of $COPC_E$s. The methodology for calculating the values is not necessarily transparent. By way of example, the report shall provide a table that displays the calculations for lead and nickel.	The reviewer is referred to Equation 4-3. A column will be added to Table 4-3 with the partition coefficients in between the SWAC and the surface water concentrations.
21	4.3.1	Wildlife Exposure Model: Looking at the values for sediment (or soil) ingestion for the various wildlife receptors in Table 3-12, we assume that the Fs value is intended to be the fraction of the diet that is soil/ sediment and that the units column should be blank. The report shall clarify/confirm this.	Tracking units of parameters in the wildlife exposure model improves transparency of the calculations. Units express important information about the basis for the value and will not be removed from Table 3-12.
22	4.3.1.2	Relative Bioavailability Adjustment Factor: For the wildlife exposure model, the 2,3,7,8-TCDD concentration was multiplied by a relative bioavailability factor (RBA) based on a study by Nosek et al. (1992). In this study, adult ring-necked pheasant hens were administered a single dose of a suspension of TCDD radio-labeled earthworms, soil, paper mill sludge, or crickets. Radioactivity remaining in the bird carcass after 24-hours was measured. This adjustment applied to $TEQ_{DF\ B}$ for sediment and soil at the shoreline, sediment outside of the western cell, shoreline background, post-TCRA shoreline, and soils north of IH-10. For tissue, this adjustment applied to $TEQ_{DF,B}$ for common rangia (site-wide and background) and blue crab (site-wide and background). Additionally, this adjustment applied to $TEQ_{DF,B}$ and $TEQ_{DF,M}$ for terrestrial invertebrates north of IH-10 and the peninsula only. It is unclear that the single exposure and uptake evaluation (after only 24 hr) utilized in the Nosek et al. study sufficiently represents reality (e.g., normal digestive tract residence time). We do not support the use of the referenced RBAs for the following reasons: a. The bioavailability study is not site-specific; b. Uncertainty regarding the dose duration and measurement time (was steady state achieved?); c. Selective uptake of TCDD in bird tissues; and d. Uncertainty in the TCDD dose concentration compared with prey/media concentrations at the San Jacinto River Site.	In the meeting with USEPA on July 18, 2012, it was agreed that additional information from the literature supporting the approach discussed in the comment will be included, and that a specific analysis of the exposure to birds without using the RBAs will be presented in the uncertainty analysis. Text will also be added to Section 4.3.1.2 to describe the uncertainty analysis.

Comment No.	Section	Comment	Response to Comment—Proposed Revision
		The referenced relative bioavailability factor shall not used, and shall be deleted from the report.	
23	4.3.1.3	Unit Conversions: Regarding the conversion of tissue concentrations expressed as wet weight to dry weight, the text shall indicate that this step was already performed (where appropriate) for each tissue sample based on the percent moisture/solids determined by the lab, and that the exposure point concentrations in Appendix C were determined after this conversion.	The reviewer is referred to Section 4.3.1.3, "Before calculating EPCs for tissue on a dry weight basis, wet weight concentrations in individual samples are first converted to dry weight concentrations using the fractional solids data for the same sample if available; if solids data is not available, the average fraction of solids data for the given species is used." Also, please note not all tissue EPCs are in dry weight, there is a mixture and that is explained by footnote a in Appendix C.
24	4.3.1.5.1	Estimating $COPC_E$ Concentrations in Plants (Concentrations of $COPC_E$s in Foods of Alligator Snapping Turtle, Killdeer, Raccoon, and Marsh Rice Rat): The full reference for the Staples et al. (1997) citation was not provided. The report shall provide this reference to the reference section.	Clarification will be provided as requested.
25	4.3.1.5.2	Estimating $COPC_E$: Concentrations in Soil Invertebrates: Soil-to-invertebrate bioaccumulation factors (BAFs) for nickel and thallium were obtained from EPA (1999b) and are provided in Table 4-9. The BAFs are presented on a wet-weight basis in the EPA reference. Because the mammalian dose calculations are performed on a dry-weight basis, it is not clear if the estimated tissue concentrations were converted to dry weight. The report shall clarify this and indicate the assumed moisture content.	Reviewers have correctly identified an error: The tissue concentration resulting from application of bioconcentration factors (BCFs) were not converted to dry weight concentrations as they should have been for nickel and thallium. A correction factor will be applied using the assumed moisture content of 84 percent in earthworms from USEPA (1993). The correction will be made to the analysis and tables will be updated to reflect revised HQs for receptors that are affected by this correction. Thallium is not a COPC for this BERA, but preliminary evaluation of soils in the south impoundment suggested that it could be a $COPC_E$ for that area. Thallium was inadvertently included in the $COPC_E$ list for this analysis. Information for this chemical will be removed from Table 4-9 and the text.
26	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Burton et al. (2006) was used to establish BAFs for estimating tissue concentrations (based on Site soil concentrations) for mercury. According to the BERA discussion and Table 4-9, an uptake factor of 3.1 was used for soil concentrations less than or equal to 1.5 mg/kg, and an uptake factor of 0.7 was used for soil concentrations greater than 1.5 mg/kg. Because these BAF values were applied to individual surface soil sample locations, the report shall add information in Appendix C that indicates the predicted CT and RM exposure concentrations for mercury for soil invertebrates.	Additional detail for the estimated CT and RM mercury concentrations in invertebrates will be added to the end of Table C-1, as requested.
27	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Regarding PCBs, the discussion indicates congener-specific models were not used to estimate invertebrate concentrations because there are no PCB congener data for soils at the Site. This is confusing because Table 4-12 indicates $TEQ_{P,B}$ values for the killdeer, Table 6-5 indicates hazard quotients for $TEQ_{P,B}$ for the killdeer, Table 6-9 indicates hazard quotients for $TEQ_{P,M}$ for the marsh rice rat and raccoon, and Table C-l indicates $TEQ_{P,B}$ and $TEQ_{P,M}$ values for soils north of IH-10. The report shall clarify and indicate how TEQ_P was evaluated for terrestrial receptors.	Text will be added to provide clarification, as requested. Dioxin-like PCB congeners were analyzed in TxDOT soils and were used to calculate TEQ_P for use in the exposure model. The text in this section is intended to convey that to evaluate exposure to total PCBs, the full suite of PCB congeners in soils was not available in order to build a congener-specific model, so a regression relationship for total PCBs using total PCBs as sum of Aroclors in soils was used as the basis for deriving an estimate of total PCBs in invertebrates. This will be clarified in Section 4.3.1.5.

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28	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: Paired soil and earthworm tissue dioxin and furan data (n = 6) from the St. Regis Paper Company Superfund Site in Cass Lake, Minnesota were used to develop a series of regression and correlation relationships for dioxin and furan congeners. These were used to estimate dioxin and furan concentrations in soil invertebrate tissue for use in the wildlife exposure model for the killdeer and raccoon. For this analysis, P-values ≤ 0.1 were considered statistically significant, and significant regression relationships between soil and tissue were developed for 11 of the 17 congeners. For the remaining 6 congeners, correlation relationships were determined with other congeners. The resulting estimated concentrations of dioxins and furans (TEQ_{DF}) in terrestrial invertebrate tissue for the raccoon or killdeer exposure scenario are shown in Table D-6. Although Sample et al. (1996) is mentioned in the discussion, there is relatively little discussion of alternative approaches. Given the small sample size and the higher than normal threshold for the determination of statistical significance, the adequacy of this approach for estimating invertebrate dioxin/furan concentrations is questionable. The report shall compare/contrast this approach generally with other relevant dioxin/furan invertebrate uptake estimates in the peer-reviewed and/or CERCLA specific literature.	Additional information from other sites or publications will be added to Appendix D, as available. Results of the evaluation of additional information will be used to qualitatively address related uncertainties.
29	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: The regression and correlation relationships developed from the Cass Lake Superfund site would not be expected to accurately predict soil invertebrate tissue concentrations at the San Jacinto River Site because the range of dioxin and furan concentrations in the six Cass Lake soil samples is much lower, especially for 2,3,7,8-TCDD and 2,3,7,8-TCDF. Additionally, the ratios between congeners in soils from the Cass Lake site are very different from congener ratios at the San Jacinto River Site. For the Cass Lake site, the highest 2,3,7,8-TCDD concentration was 1.83 ng/kg, and the highest 2,3,7,8-TCDF concentration was 11.3 ng/kg (Table *D-1*). In contrast, at the San Jacinto River Site, the highest soil 2,3,7,8-TCDD concentration was 8,650 ng/kg, and the highest 2,3,7,8-TCDF concentration was 20,600 ng/ kg (Table 6-17 in the Preliminary Site Characterization Report). According to Appendix D, the 2,3,7,8-TCDD congener was not detected in 5/6 of the Cass Lake earthworm samples. In the one sample where 2,3,7,8-TCDD was detected in tissue, it was not detected in soil. Because no statistically significant relationship between soil and earthworm concentrations was identified for some congeners, a correlation approach was used, which compared the ratio of congener concentrations in earthworm tissue. The ratio between concentrations of 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD was used to predict the 2,3.7,8-TCDF concentration in invertebrate tissue. For the Cass Lake site, the average 1,2,3,6,7,8-HxCDD concentration in soil was about 50 times greater than the concentration of 2,3,7,8-TCDF in soil. In contrast at the San Jacinto River Site, the average TCDF concentration in Area 3 soils was over 3,200 times the average 1,2,3,6,7,8-HxCDD concentration in soils (Table 6-17 in PSCR). This suggests that the use of the Cass Lake soil data will greatly underestimate the concentration of TCDF in invertebrate tissue at the San Jacinto River Site. Given the significant difference in soil concentrations for TCDD and TCDF, and the uncertainty associated with the ratio approach, the adequacy of this approach for	The overall range of concentrations in the “soil” dataset for the Site is highly skewed by the samples collected from within the original 1966 perimeter of the impoundments north of I-10. Similarly, the dioxin and furan fingerprints of soils within this area, which are characterized by large fractions of the tetrachlorinated congeners, are substantially different from those of soil samples collected outside the impoundments. The concentration ranges of 2,3,7,8-TCDD and 2,3,7,8-TCDF are greater in the impoundments north of I-10 than in Cass Lake soil. However, the ranges and central tendencies of concentrations of these congeners outside of the impoundments are quite similar to those of 2,3,7,8-TCDD and 2,3,7,8-TCDF in Cass Lake soils. This will be clarified in Appendix D. The ratios of maximum values of TCDF to TCDD in Cass Lake soil is 6; in the San Jacinto data, this ratio is approximately 3 for soils both inside and outside of the impoundments. Ratios of congeners for both the Cass Lake and SJRWP data sets inside and outside of the impoundments will be more clearly described in Appendix D. 2,3,7,8-TCDD was not detected in four of six earthworm tissue samples (U-qualified) and was estimated in one sample (J-qualified). In Cass Lake soil, 2,3,7,8-TCDD was not detected in three samples, and was estimated in two. Nevertheless, a significant regression relationship was derived for this congener. Uncertainties associated with the censored data for 2,3,7,8-TCDD will be discussed in revisions to Appendix D. We agree with the comment that 2,3,7,8-TCDF and 1,2,3,6,7,8-HxCDD ratios for soils within the impoundment are high relative to these ratios in the Cass Lake data set. Therefore, significant correlations with another congener having concentrations more similar to TCDF and with a significant regression relationship will be applied to predict TCDF concentrations in soils within the 1966 impoundment perimeter. Revisions will be made to Appendix D to describe these changes in the analytical approach and results. Consistent with response to comment 28, additional discussion will be provided regarding the appropriateness of this approach.

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		estimating invertebrate dioxin/furan concentrations is questionable. The report shall compare/contrast this approach generally with other relevant dioxin/furan invertebrate uptake estimates in the peer-reviewed and/or CERCLA specific literature.	
30	4.3.1.5.2	Estimating $COPC_E$ Concentrations in Soil Invertebrates: There is a statement in Section 2.1 of Appendix D that "the ranges of dioxin and furan concentrations in soil at the Cass Lake site were similar to the range of concentrations in soils at the San Jacinto River site." This shall be revised. The total TEQ ranges may be similar, but the individual congener ranges were not.	Clarification will be provided as requested; please see response to Comment 29 indicating that a more specific analysis of soils inside and outside of the impoundment will be added to discuss relationships with the Cass Lake soil data set.
31	4.3.1.6	Wildlife Exposure Units: Figure 4-9 depicts the exposure areas and samples used for the killdeer evaluation. The report shall explain why all of the area on the west side of the upland sand separation area was used for the assessment when surface soil data was not available for the far western third of the property. Additionally, the report shall state whether this inclusion was conservative.	Exposure areas were determined by considering the areas that constitute appropriate and accessible habitats. The exposure area selected reflects all available data, because the area use factor (AUF) is set to 1 for this receptor, per Table 4-11, which is the most conservative possible approach. Clarification will be provided as requested.
32	4.3.1.6	Wildlife Exposure Units: Figure 4-10 depicts the exposure areas and samples used for the raccoon evaluation. Very limited soil/sediment data was available for these areas, and clams and small fish were not collected in this area. The report shall explain why all of the area along the west shoreline of the Southern Impoundment and along the eastern shoreline on the land mass across the Old River Channel (and south of IH-10) was used for the assessment. Additionally, the report shall state whether this inclusion was conservative and how will it be integrated with an ecological assessment for the Southern Impoundment.	The raccoon is not a receptor for the south impoundment; the SLERA for the south impoundment is presented in Appendix E. To address risks to raccoon for the northern impoundments and surrounding aquatic environment, shoreline sediments and the tissue samples of aquatic biota within the exposure area for the impoundments north of I-10 and surrounding aquatic habitats were included. In preparation of the sampling and analysis plans, the sediment and tissue sampling designs explicitly considered the risk assessments, and these considerations are described in the data quality objectives provided in Section 1 of each of those two sampling and analysis plans (SAPs). Based on the results of surface sediment samples collected in the Old River to date, concentrations in beach sediments there would be expected to be low. Therefore, the sediment and tissue samples used in the BERA, as a whole have a conservative bias, because they are focused in areas adjacent to the source material that was in the aquatic environment north of I-10 at the time of sampling.
33	4.3.1.6	Wildlife Exposure Units: Similarly, Figure 4-11 depicts the exposure areas and samples used for the great blue heron, spotted sandpiper, and marsh rice rat evaluations. Very limited sediment data was available for the areas south of IH-10, and clams and small fish were only collected in an area along the east side of the river channel shoreline (and south of the IH-10 bridge). It is not clear how data from these areas will be incorporated into the exposure calculations. The report shall clarify this. Additionally, the report shall state whether this inclusion was conservative and how will it be integrated with an ecological assessment for the Southern impoundment.	The exposure area selected reflects all available data, because the AUF is set to 1 for these receptors, per Table 4-11. This is the most conservative possible approach. Clarification will be provided as requested. The reviewer is referred to the second paragraph of the Introduction (Section 1), which addresses the ecological risk assessment process for the south impoundment area. This process begins with the SLERA, which is presented in Appendix E (to be modified according to comments 74 and 75). The approach was developed consistent with the conceptual site models (CSMs) for the site, updated in the Preliminary Site Characterization Report (PSCR).

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34	4.3.1.7	Calculation of Exposure Point Concentrations: Appendix C shall be amended to include the surface water CT and RM exposure point concentrations for TEQs and Total PCBs that were used for determining the bird dose (i.e., surface water ingestion).	Details for $TEQ_{DF,B}$ will be provided in Appendix C as requested. Because PCBs are bioaccumulative, animals receive the majority of ingestion exposure by eating aquatic species, and the contribution of PCBs in ingested water to the total dose of PCBs to birds is expected to be very low. Given that predictions in water are highly uncertain and PCBs have very low solubility, ingestion of waterborne PCBs was not considered.. This will be clarified in Section 4.3.1.7.
35	4.3.1.9	Results: The text states that the results of calculations using BAFs and regression models for invertebrates and plants were not tabulated, but were incorporated directly into the wildlife exposure model. For transparency, this particular part of the dose calculation shall be presented along with the corresponding soil/sediment exposure point concentration.	The BAFs and regression models are provided in Table 4-8; as described in the text, these are multiplied by the appropriate soil or sediment EPC to generate an estimate for plants and invertebrates. The resulting EPCs for invertebrates and plants will be added to Appendix C for transparency.
36	4.3.1.9	Results: Table 4-12 presents the final estimates of the daily ingestion rate of each $COPC_E$ for each receptor. We were not able to duplicate the values indicated for the raccoon. The report shall confirm/clarify the calculations. This may be related to uncertainty associated with the exposure areas assumed for the raccoon (i.e., see comment 9). 7/15/2012 Correction: i.e., see comment 67.	Please see response to comment 67, the table will be corrected to show exposure assumptions accurately, and this will correct the discrepancy. The table will be corrected.
37	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The linear regression models for each congener or homologue group from Elliott et al. (2001) were used to estimate egg concentrations for the blue heron, cormorant, and sandpiper. The regression equations are shown in Table 4-13. Levels of 2,3,7,8-TCDF were not linearly related for fish and egg concentrations ($p = 0.07$). The report shall discuss the uncertainty associated with the use of the Elliot, et al. (2001) model for this congener.	Additional information will be provided in Section 7.2.2.1 to describe uncertainty associated with the fish-to-egg model.
38	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The discussion on page 4-29 explains that for the fish-to-egg calculations, an individual sample of each medium was used to represent the CT and RM exposures. The sample selected was that with the $TEQ_{DF,B}$ concentration closest to the calculated CT or RM for the particular exposure unit. The report shall provide more discussion on why this calculation method was selected and the location, sample number, and congener and homologue concentrations of the individual samples selected for use. Additionally, this discussion states that it was considered overly conservative to use the CT and RM for each congener to estimate the concentrations of dioxins and furans in bird eggs. The report shall explain this statement.	Clarification will be provided in the discussion of model implementation in Section 4.3.2.1.2.

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39	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): The results of the TEQ calculations using the regression models to estimate concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are shown in Table 4-15. For transparency, the report shall show the step-by-step calculation of the values in Table 4-15 for the combinations that follow. This would include presentation of the individual congener concentration EPCs (in food and sediment) as inputs to the calculation. a. Cormorant/TCFD/prey only/CT/TEF_{max}; b. Heron/PeCDD/prey + sediment/RM/TEFmin; c. Sandpiper/∑HxCDF/prey + sediment/CT /TEF_{min}.	An exhibit will be added detailing step-wise calculation of each of the requested combinations.
40	4.3.2.1.2	Implementation of the Prey-to-Egg Model (Estimated TEQ Concentrations in Bird Eggs): It appears that the TEF /TEQ values are missing for the heron and sandpiper (Table 4-15, background: prey + sediment). The report shall provide these values or explain why they were not presented.	Upstream data for shoreline sediments was inadvertently overlooked in calculation of background exposures of heron and sandpipers to dioxins and furans. The analysis will be revised and details added to Table 4-15. For estimation of egg PCBs, with consumption of prey and sediment, only the cormorant was evaluated because there is no background PCB data for shoreline sediment. The reviewer is referred to the last bullet in Section 4.3.2.3.
41	4.3.2.2.1	Overview of Literature Found (Estimating PCB Concentrations in Bird Eggs): The complete reference for Naito and Murata (2007) was not provided in the list of references. The report shall add this to the list of references. Additionally, the actual BMFs (biomagnification factors) in this paper were cited from other papers.	The appropriate citation will be added.
42	4.3.2.2.1	Overview of Literature Found (Estimating PCB Concentrations in Bird Eggs): The results of the TEQ calculations using the indicated BMFs (Table 4-16) to estimate PCB concentrations in eggs of the neotropic cormorant, the great blue heron, and the spotted sandpiper are shown in Table 4-17. For transparency, the report shall show the step-by-step calculation of the values in Table 4-17 for the combinations that follow. This would include presentation of the individual PCB congener concentration EPCs (in food and sediment) as inputs to the calculation. a. Cormorant/PCB l05/prey + sediment/CT; b. Cormorant/PCB126/background: prey + sediment/RM; c. Heron/PCB 077/background: prey/RM; d. Sandpiper/PCB 118/prey only/CT.	An exhibit will be added detailing step-wise calculation of each of the requested combinations and is compiled in the document following the figures.

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43	4.3.2.3	Egg Exposure Scenarios: Previous sections detail the approach for estimating egg TEQ_{DF} and TEQ_{P} concentrations using regression equations or BMFs applied to empirical fish tissue concentrations. This information is needed to evaluate potential risks to birds by comparing estimated TEQ concentrations in eggs to TRVs expressed as egg concentrations (wet weight). Exposure scenarios detailed here reflect an evaluation of egg concentrations resulting from combinations of prey (fish, crabs, or common rangia) and sediment. The report shall provide clarification regarding how egg tissue concentrations were estimated based on uptake from sediment, crabs, and common rangia. This is not clear.	Clarification was provided by reviewers during in a meeting with USEPA on July 18, 2012, as follows: There is uncertainty associated with using a model derived from data in which fish tissue is the independent variable in an application in which clam and crab tissue and sediment are combined to represent the independent variable. This uncertainty should be discussed in Section 7. Additional discussion will be added to Section 7 to address this uncertainty.
44	4.4.2	Derivation of Parameter Distributions: Table 4-19 displays the distribution characteristics for the various exposure parameters used in probabilistic risk analysis. The report shall discuss why any particular reference (e.g., DREBWQAT (1999) and Fernandes (2011)) was used here, and not in the initial dose calculations. Also, the report shall explain a triangular distribution.	The probabilistic risk assessment necessitates the use of not only a central tendency, which is consistent with the deterministic risk assessment references, but also a measure of variance and range, which are not contained in the deterministic risk assessment, hence the use of these additional references to provide these statistics. Clarification will be provided as requested.
45	5.3	Benthic Macro-invertebrate Communities: Notes f, h, and i are missing from Table 5-1. This table shall be revised to include these.	The table will be corrected. The same footnotes are missing from Table B-13, which is the duplicate benthic TRV table in Appendix B, and this table will be corrected as well.
46	5.3	Benthic Macro-invertebrate Communities: The marine chronic criterion for lead (Texas Surface Water Quality Standards (TSWQS), §307.6 (c)) of 5.3 ug/L shall be used for evaluating estimated pore water concentrations as this value is more conservative that the federal criterion. This is an ARAR (Applicable or Relevant and Appropriate Requirement).	The status of a benchmark as an ARAR is not a consideration in the selection of TRVs in a risk assessment. Moreover, the AWQC for lead was not needed for the risk assessment for benthic invertebrates, because values that could be used to evaluate risk to the benthic invertebrate community using the primary line of evidence, bulk sediment concentrations, were available. The surface water criterion for lead will be removed from Table 5-1 and from Table B-13.
47	5.3	Benthic Macro-invertebrate Communities: For the evaluation of reproductive risks for molluscs, the BERA used the paired NOAEC/ LOAEC (no-observed adverse effect concentration/lowest-observed adverse effect concentration) values of 2 and 10 ng TCDD/kg ww tissue, respectively, for delayed gonadogenesis in males (Wintermyer and Cooper (2007). An NOAEC of 2 ng TCDD/kg ww tissue is too high given that this concentration has been found to adversely affect early stages of oyster gametogenesis (Wintermyer and Cooper (2007) and veliger larval survival (Cooper and Wintermyer (2009). The report shall be revised to include the 2 ng TCDD/kg ww tissue concentration as the LOAEC, and a lower NOAEC shall be determined based on an appropriate literature value.	The TRV addressed by this comment is only appropriate for evaluation of risk to bivalve molluscs, as explained in Appendix B, because studies with other types of benthic macroinvertebrates have demonstrated that several macroinvertebrate taxa are not sensitive to 2,3,7,8-TCDD toxicity. The concentration of 2 ng TCDD/kg tissue will be considered the LOAEL in the revised BERA. There is no information to support identification of a corresponding NOAEL. However, we do not agree with Cooper and Wintermyer (2009), which cites Wintermyer and Cooper (2003) to support a conclusion that 2 ng/kg TCDD in eastern oysters (*Crassostrea virginica*) causes reduction in veliger larval survival. Wintermyer and Cooper (2003) placed their wild-caught test subjects in Newark Bay, in Arthur Kill of the Raritan Complex, and in a reference area (Sandy Hook), all in New Jersey. While Wintermyer and Cooper (2003) document the presence of TCDF and PCBs in adult oyster

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			tissue, they do not report on contamination of this area with metals, PAHs, and estrogenic compounds, all of which are common in urban estuaries and all of which could have affected the endpoints evaluated. Additional discussion of Wintermyer and Cooper (2003) will be provided in Appendix B and Section 5.3. The discussion of risk to bivalves in Section 6.2.3 will also be expanded to better explain uncertainties. Because the most technically robust evidence from this body of work is for a histological endpoint, and effects at the lowest concentrations were marginal in females and did not occur in males at the lowest dose, a NOAEC below 2 ng/kg tissue will not be proposed.
48	5.3	Benthic Macro-invertebrate Communities: Continuing with a discussion of the NOAEC/LOAEC values for molluscs, the referenced studies only dosed the molluscs with 2,3,7,8-TCDD, whereas the molluscs at the site are potentially exposed to all of the dioxin and furan congeners. Thus, site molluscs would have a greater exposure to total dioxins/furans overall. This compounds the uncertainty associated with the selected tissue residue endpoint for molluscs. The report shall evaluate/clarify this.	As described in Appendix B and in Attachment B2 to Appendix B of the RI/FS Work Plan, invertebrate cells do have aryl hydrocarbon receptor homologues, but these do not bind dioxin. Therefore, the toxicity of 2,3,7,8-TCDD is not necessarily an indication of toxicity of other 2,3,7,8-substituted dioxin and furan congeners as it is in vertebrates. Consequently, it is not appropriate to make any assumptions about compounding uncertainty. The draft BERA clearly states in Section 6.2.5 that exposure of bivalves to other dioxin and furan congeners cannot be interpreted due to a lack of toxicity information. Text will be added to Section 7, the uncertainty analysis, to highlight this data gap.
49	5.4	Fish: For nickel, the results of tests with marine fish were combined to determine a chronic TRV for nickel expressed as a concentration in water (3,600 ug/L; Table 5-2 and Table B-16). The marine chronic criterion for nickel (TSWQS, §307.6 (c)) of 13.1 ug/L shall be used. This is an ARAR.	The status of a benchmark as an ARAR is not a consideration in the selection of TRVs in a risk assessment. The nickel TRV for fish was well-considered and represents a range of marine fish species. The value presented in Tables 5-2 and B-11 is a conservative representation of no observed adverse effects concentrations for marine fish from the peer reviewed literature and USEPA's water quality criteria document for nickel. The derivation of the TRV is discussed in Section 3.9.1 of Appendix B. In light of the available information describing the actual toxicity of nickel to marine fish, it would be inappropriate to suggest that a value 200 times lower than the geometric mean of several NOAECs is a toxicity threshold. No change will be made. Tables 5-2 and B-11 will be corrected to show this value as a NOAEC.
50	5.4	Fish: The TRVs (NOAEL and LOAEL fish whole body concentrations) for Total PCBs are summarized in Tables 5-2 and B-11 and are discussed in Sections 2 .2 .1.1 and 2.2.1.2 of Appendix B. These TRVs are largely based on studies where fish were exposed to Aroclor 1254 and tissue was analyzed for Total PCBs. The report shall briefly discuss the uncertainty associated with the use of Aroclor toxicity data relative to the congener tissue data used for the BERA.	Additional discussion requested will be provided in Section 7 and in Appendix B.

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51	5.4	Fish: Regarding the TCDD TRV (from Steevens et. al. (2005)), our understanding is that the tissue residue TRV is based on concentrations in fish eggs and embryos rather than whole fish. The report shall clarify this. It appears that whole fish concentrations are used in the hazard quotient calculations (Section 6.3.4).	Clarification will be provided in the text of Section 5.4 and in Appendix B.
52	5.6	Birds and Mammals: The avian and mammalian TRVs for Total PCBs are summarized in Tables 5-3, 5-4, and B-11, and are discussed in Sections 2.2.3 and 2.2.4 of Appendix B. These TRVs are largely based on studies where birds or mammals were exposed to Aroclor 1254 in their diets. The report shall briefly discuss the uncertainty associated with the use of Aroclor 1254 (primarily) toxicity data relative to the total PCB (sum of Aroclors) tissue and sediment data used for the BERA.	Clarification will be provided in Section 7 and in the text of Appendix B.
53	5.6	Birds and Mammals: The report shall re-evaluate the calculated NOAEL and LOAEL values for the avian TRVs for barium. We were not able to duplicate the values indicated in Table 5-3 based on the text in Section 3.2.2 of Appendix B. The report shall also evaluate the indicated TRVs. Presumably this would be relevant for the SLERA for the area south of IH-10 because barium is not a $COPC_E$ for wildlife receptors for the area north of IH-10.	Inclusion of a discussion of a barium TRV for birds is a mistake, and related information will be removed from the report and Appendix B. If barium is a $COPC_E$ for birds in the south impoundment area, TRV calculations will be checked as requested.
54	6.2	Risks to Benthic Macro-invertebrate Communities: This discussion generally compares the various screening values with the bulk sediment or estimated pore water concentrations, indicates the number of exceedences, and plots the sample locations on a series of figures. This discussion shall be revised to indicate the concentrations (i.e., bulk sediment or estimated pore water) that exceeded the screening values.	The additional information requested will be provided in the maps cited in Section 6.2 which show results for those locations where concentrations exceed the TRV or screening value.
55	6.2.3	TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs: Potential risks associated with critical tissue residue in molluscs shall be reevaluated given the concerns regarding the selected tissue NOAEC/LOAEC values.	The discussion will be modified. Please see response to comment 47.
56	6.2.3	TCDD in Clam Tissue Relative to the Critical Tissue Residue for Molluscs: Absent confirmation sampling, it is unknown whether risks to molluscs in the vicinity of Transect 3 have been greatly reduced as a result of the TCRA. The report shall clarify this.	The reviewer is referred to the first sentence in the last paragraph of Section 6.2.3 which states: “It is not possible to evaluate post-TCRA risk to clams in the vicinity of Transect 3…” Clams were collected directly along the shoreline of the wastes from the northern impoundments, along Transect 3, as shown in Figure 4-1. This area is clearly within the TCRA footprint. The text will be modified to clarify the basis for the statement suggesting that the TCRA has affected exposure and risk to clams.
57	6.2.3/8.1	The conclusion that risks to bivalves are low in transects 3 and 5 based on the available data on clam tissue is not appropriate.	The document does not conclude that baseline (pre-TCRA) risks to molluscs from Transect 3 (adjacent to the impoundments) are “low.” The document acknowledges some reproductive risk to

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		If the assertion is that the TCRA has addressed the affected bivalves near the pits, monitoring post-TCRA will be necessary along with appropriate action levels in clam tissue.	*individual* bivalves in the area adjacent to the northern impoundments, stating in Section 6.2.3 that "individual clams from the area represented by Transect 3, assuming they are as sensitive as the oysters of Wintermyer and Cooper (2007), are at risk of reproductive impairment ". However, the assessment endpoint is "stable or increasing *populations* of bivalves." The possible effect on this assessment endpoint due to concentrations in bivalve tissue above the effects thresholds identified in samples from Transect 3 is unknown, because the spatial distribution of molluscs with those concentrations is unknown. However, data presented in the PSCR indicated that clam tissue concentrations of TCDD are somewhat correlated with sediment concentrations. Because very high concentrations of TCDD in sediments have limited distribution on the site, it is reasonable to conclude that effects on individuals are correspondingly limited, and that therefore, entire populations of bivalves in the site as a whole are not at risk. The text of Sections 6.2 and 8.1 will be revised to better convey the difference between risks to individuals and risks to the assessment endpoint. Concentrations of TCDD in three of five clam tissue samples from Transect 5 are below the lowest threshold of effects on molluscs. The effect indicated by the TRV is a histological abnormality, which is presumed to lead to some unspecified reproductive effect. The assessment endpoint addressed by clam tissue is "stable or increasing populations of bivalves." The conclusion that risks are low to bivalve populations because of slight exceedance of a histological effects threshold in less than half of the samples at Transect 5 is appropriate and will not be changed. Please note that the statements in the last paragraph of Section 6.2.3 do not assert that the TCRA has addressed risks to bivalves. It presents information that informs but does not attempt to resolve the post-TCRA risk condition.
58	6.2.5	Summary: Lines of Evidence for Benthic Macro-invertebrate Communities: The actual risk to populations of molluscs (based on tissue concentrations of dioxins/furans) is unknown. Additionally, consideration of potential risks to molluscs directly adjacent to the impoundment or elsewhere on the Site will be driven by the selected tissue NOAEC/LOAEC (see comments for Section 5.3). The report shall clarify this.	Please see response to comment 57.
59	6.3.1	Estimated Concentrations of Metals in Fish Diets Relative to TRVs: Hazard quotients for fish exposed to cadmium, copper, mercury, and zinc in foods and sediment are summarized in Table 6-3 and indicate that the LOAELs are not exceeded. These hazard quotients will be revisited based on the report revision in response to comment 8. 7/15/2012 Correction: i.e., see comment 65	Table 6-3 shows HQ calculated using NOAELs an LOAELs, all of which are below 1. The table and related conclusions will not be revised.
60	6.3.2	Estimated Concentrations in Surface Water Relative to TRVs: A hazard quotient of less than 0.1 was determined for fish exposed to nickel in surface water (Table 6-4). The hazard quotient will be above one using the chronic Texas criterion (see previous comment 39). The report shall be revised to include the chronic Texas criterion.	Please see response to comment 49.

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61	6.3.3	Total PCB Concentrations in Whole Fish Relative to the TRV for Fish: See previous comment 40 regarding the toxicity studies used to derive the fish whole body TRVs.	See response to comment 40.
62	6.3.5	Summary - Lines of Evidence for Fish: This discussion concludes that overall, risks to fish on the Site are negligible. This conclusion will be revisited based on the report revision in response to previous comments regarding the exposure concentrations (surface water), diet, and TRVs for fish.	Conclusions about risks to fish will not be revised on the basis of changes resulting from previous comments.
63	(7.)4.2.5	Datasets Used to Evaluate Exposure to Fish: The references for the killifish movement/home range were not provided in the reference section. The report shall provide the full references.	It appears that the commenter is referring to Section 4.2.5. The requested references will be added.
64	8.2	Characterization of Risks to Fish: The discussion summarizes that baseline risks to the assessment endpoints (stable or increasing populations of benthic omnivorous fish, benthic invertivorous fish, and benthic piscivorous fish on the Site) arc negligible. This conclusion will be revisited upon the report revision in response to previous comments regarding the exposure concentrations (surface water), diet, and TRVs for fish.	Please see the response to comment 62.
65	4.2.6 Corrected 7/15/2012	Results of Fish Exposure Assessment: The values in Table 4-6 shall be related with the exposure point concentrations in Appendix C, if applicable. If not applicable, the report shall explain how these weighted concentrations were derived and indicate where the data is summarized so this can be verified. Finally, the report shall clarify why is the total diet (last column in Table 4-6) simply the sum of each of the CT and RME values. Have the individual values for each food type already been modified by the proportion each food type represents in the diet?	Section 4.2.1 describes calculation of weighted fish diets. This section explains that the EPCs for each component of the fish diet, as expressed in Appendix C, are multiplied by the relative proportion of that item in the diet of the fish that is outlined in Table 4-2. The reviewer is referred to Equation 4-4, which provides the explanation of how the total diet is calculated; the values in Table 4-6 are summed to provide the total diet in the last column. This will be clarified by providing footnotes to Table 4-6 that describe this process.
66	8.6	Ecological Risk Assessment Conclusions: The overall risk assessment conclusions will be revisited after receipt of a revised BERA and accompanying responses to agency comments.	According to a discussion with USEPA and other reviewers on July 18, 2012, the draft final report will be submitted with a redline/strikeout of the text, to facilitate the USEPA's final review.
67	4.3 Corrected 7/15/2012	Exposure of Reptiles, Mammals, and Birds: Table 4-7 presents the exposure areas and assumptions for food/sediment/soil for various receptors. The exposure assumptions for the raccoon were a bit confusing. Presumably, concentrations in molluscs for the peninsula shoreline were used. It was not clear why this was not the case	This comment highlights two errors that will be corrected: In Table 4-7, the "Terrestrial Invertebrates" cell for Raccoon will be revised to state "BAFs from peninsula soils." The cell for "Benthic Invertebrates" will be revised to remove the reference to use of a BAF, because empirical tissue data were used in the exposure model for raccoon.

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		for small fish also since exposure point concentrations were presented for this subset in Appendix C. For terrestrial invertebrates and plants, it was unclear why concentrations were modeled from soil concentrations for soils north of IH-10 if soil ingestion was modeled for the entire peninsula. The report shall clarify/explain these issues.	In addition, empirical tissue data for clams and small fish were incorrectly used in the model from the entire aquatic area instead of restricting to the peninsula exposure area as correctly described by Table 4-7 and Figure 4-10. These calculations will be revised and dose and HQ tables will be updated as appropriate.
68	Table 4-5	Were there not 10 samples collected and analyzed from each FCA? The report shall clarify this.	Although 10 catfish fillet samples were collected from each fish collection area (FCA) for evaluation of human health risks, for ecological risk endpoints, a total of 10 whole catfish and 10 whole killifish were collected across the entire site. The sampling emphasized FCA2, which contains the area of the northern impoundments. This is the design described in the Sampling and Analysis Plan: Tissue Study (2010b). Clarification will be provided in Section 4.2.3.
69	Table 5-1	The report shall provide additional information supporting the assumption of dividing the LC_{50} by 10 results in a defensible estimation of the NOAEC. In this table, an uncertainty factor of 10 is applied to a LC_{50} resulting in a NOAEC and an EC_{50} yielding both a NOAEC and LOAEC. There is a disconnect in the logic in using this factor.	The reviewer is referred to Section 1.3 of Appendix B of the BERA, which describes the use of uncertainty factors. This information will be summarized in Section 7 of the BERA.
70	Table 5-1	This table has an incorrect reference for the TCDD value. The comment indicates that the range was derived from table B-5, but it should be Table B-4. The report shall be revised to correct this.	The correction will be made.
71	Table 5-2/ B-14	The TCDD value is described as a NOAEC; however, the source of this value indicates that it was the geometric mean of the NOER and LOER. The report shall either provide justification for the designation as a NOAEC or rename.	Clarification will be provided.
72	Table 5-2/ B-14/B-11	The source of the NOAEC and LOAEC for PCBs in fish is not clear. Although a summary of the studies used to derive these values is included in Section 2.2.1.1, 2.2.1.2, and Appendix B, it was not clear which of the studies were selected and which were not to calculate the NOAEC and LOAEC in this BERA. The report shall provide a table similar to B-4 for fish, and include only those studies used to calculate the TRVs.	A table will be prepared that shows the studies compiled to develop the PCB TRV for fish and clearly indicates which studies were selected for calculation of the NOAEL and LOAEL TRVs.
73	Figure 2-2	This figure combines the worker and trespasser receptor categories. Additional clarification/justification shall be provided for why these categories should be combined.	The CSM figures are for the site overall. Human receptors are not addressed by the BERA. Additional detail on the human health risk evaluation will be presented in the Baseline Human Health Risk Assessment.
Comments on Appendix E: Draft Screening-Level Ecological Risk Assessment, South Impoundment			
74	2.5	Assessment Endpoints: In Table E-3 (assessment endpoints), the assessment endpoint for mammals does not pair up with the selected receptor (pocket gopher) because it is an herbivorous mammal. The report shall include an omnivorous mammal (e.g., shrew, marsh rice rat, or armadillo) and revision of Table E-3.	The omnivorous mammal that will be added to the receptors evaluated by the ecological risk assessment for the south impoundment area is the opossum (*Didelphis virginiana*). The marsh rice rat is more appropriate for evaluation of aquatic exposures, and the shrew and armadillo would not be expected in habitats like that provided by the south impoundment area.

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75	3.2	Ecological Risk-Based Screening Methods: For the semi-volatile organic chemicals (SVOCs), footnotes shall be added to Table E-5 to indicate where the median value for the Site-specific background concentrations was used. Additionally the explanation for note c is unclear (also in Table E-6). The report shall clarify this.	The Site-specific background concentrations were inadvertently placed into a column suggesting that they are Texas median background concentrations. Also, the SVOC values should have been bold, indicating that they were used for screening in Table E-6, as stated in the existing footnotes. These errors will be corrected. Footnotes will be clarified.
Comments Dated February 7, 2013, on Draft-Final Baseline Ecological Risk Assessment Dated August 2012			
1	3.4.4	The full citation for the Shields 2012 reference (related to the brown pelican range) is not provided in the list of references. This citation shall be added.	The citation will be added to the reference list.
2	3.4.4	There is a typographic error in the last sentence of Section 3.4.4. The reference to Section 4.1.3.6 shall be revised to state Section 4.3.1.6.	The error will be corrected.
3	Table 4-8	The table was not revised to indicate that the raccoon's fish dose was modeled for the peninsula fish only as was stated in the response to comment number 67; it currently states "site wide". The table shall be revised to include this.	The table will be revised accordingly.
4	5.3, 6.2.3; Table B-4	Laboratory studies in Wintermyer and Cooper (2003) are relevant to these sections. In addition to the reproduction studies of the oysters transplanted to impacted field locations in New Jersey, Wintermyer and Cooper injected (laboratory) adult oysters with tritium-labeled TCDD, and these were strip spawned after 28 days of exposure. Eggs from each treatment group were fertilized with sperm from the corresponding treatment group. The nominal concentrations were 2.0 and 20 pg/g and the concentrations in tissue were reported as 0.966 and 27.7 pg/g TCDD. For both treatment groups, there was a reduction in the number of veliger larvae compared to controls. For the 2.0 pg/g treatment group, roughly half of the eggs were fertilized, and of those, there was 100% mortality within 48 hours. This lab study indicates a tissue LOAEC for impaired reproduction and reduced larval survival as low as 1 pg/g or 1 ng/kg. The BERA shall be revised to address this result. In an email from USEPA on April 19,2013: EPA agrees that the LOAEC value from Wintermeyer and Cooper (2003) included in the draft report is correct, and therefore retracts Comment #4 (Miller 2013, Pers. Comm.).	Although the original comment incorrectly interprets the dosing regime in the laboratory component of Wintermyer and Cooper (2003), the comment correctly states that the laboratory component of this study is relevant, and that the lower dose (2 ng/kg) in oyster tissue in the laboratory study resulted in reduced egg fertilization and reduced larval survival in oysters. The text of the BERA will be revised to address the potential for these effects in oysters with tissue concentrations of 2 ng/kg ww or greater. In addition, earlier text of responses require correction. For accuracy, the following parts of the response to earlier comment 47 (above) are retracted: Comment 47: However, we do not agree with Cooper and Wintermyer (2009), which cites Wintermyer and Cooper (2003) to support a conclusion that 2 ng/kg TCDD in eastern oysters (*Crassostrea virginica*) causes reduction in veliger larval survival … …Because the most technically robust evidence from this body of work is for a histological endpoint, and effects at the lowest concentrations were marginal in females and did not occur in males at the lowest dose... Similarly, the italic text in the following replaces the corresponding wording in the ninth sentence of the original response to earlier comment 57 (above): Comment 57: The effect indicated by the TRV *indicates* histological abnormaliti*es (Wintermyer and Cooper 2007), and reproductive effects including reduced fertilization success and reduced larval survival (Wintermyer and Cooper 2003).*
5	7.2.2.1	"PCBs" shall be removed from the title for this section because PCBs are not discussed there.	The section heading will be corrected.

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6	References	The link provided in the reference section for the U.S. Environmental Protection Agency paper on dioxin bioavailability (USEPA, 2010b) is incorrect and shall be revised.	The reference for USEPA (2010b) was inadvertently run in with USEPA (2012), making it appear that the link was incorrectly associated with the former reference. The two references will be separated with a line break, and a link will be will provided for the former reference.
7	References	The full citations for U.S. EPA (1986) and WHO (2001) in Table 5-1 were not carried forward to the reference list. These shall be added to the reference list.	These references will be added to the list.
8	Table B-12	This table lists the data used to develop fish tissue-based toxicity reference values (TRV). Suitable data references should be judged on criteria including sensitive life stage, chronic exposures, a protective endpoint (mortality is not very protective), species representative of the site receptors, and evaluating PCBs as a mixture (because that is what the site exposure would be). The following data sources are not appropriate for the reasons given, and shall not be included in the TRV derivation: a. Duke et al 1970 (only examined acute exposures); b. Lieb et al 1974 (used rainbow trout, a coldwater species not representative of Gulf of Mexico fish); c. Nestel and Budd 1975 (used rainbow trout, a coldwater species not representative of Gulf of Mexico fish); d. Mauck et al 1978 (used brook trout, a coldwater species not representative of Gulf of Mexico fish); e. Berlin et al 1981 (used lake trout, a coldwater species not representative of Gulf of Mexico fish); f. Mac and Seelye 1981 (used lake trout, a coldwater species not representative of Gulf of Mexico fish); and g. Powel et al 2003 (used chinook salmon, a coldwater species not representative of Gulf of Mexico fish). Instead, the following data sources shall be included in the TRV derivation: a. Orn et al 1998 ("The Impact on Reproduction of an Orally Administered Mixture of Selected PCBs in Zebrafish (Danio rerio)"); LOAEL 2.7 mg/kg. b. Westin et al 1983 ("Effects of Parental and Dietary PCBs on Survival, Growth, and Body Burdens of Larval Striped Bass"); NOAEL 3.1 mg/kg. In an email from USEPA on April 19,2013: Comment #8: EPA agrees that the NOAEL value from Westin et al. (1983) should be 4.4 mg/kg instead of 3.1 mg/kg, and therefore revises the comment to include the 4.4 mg/kg value in the TRV derivation. (Miller 2013, Pers.Comm.).	The studies of PCB toxicity to fish were selected based on criteria described in Section 1 of Appendix B. Moreover, the use of several salmonid species in calculation of the TRV is conservative because salmonids tend to be among the most sensitive fish taxa to many toxicants, including PCBs. Note that chinook salmon has both freshwater and marine life stages. Although we would not anticipate that PCB toxicity would be different in freshwater fish than in marine or estuarine fish, the changes will be made, as requested. However, there are important uncertainties associated with both studies that USEPA has decided to include. The effect on the final TRVs for total PCBs is to make them highly conservative. Orn et al. (1998) evaluated effects on individual organs in fish, requiring dissection and removal of the ovaries and liver. "Whole body" concentrations of total PCBs were measured after these organs were removed. Because PCBs concentrate in these organs, the authors acknowledge that the resulting "whole body" concentrations are biased low. Moreover, these authors used a selection of 20 PCB congeners, resulting in a mixture that is not commonly found in nature. The lack of representativeness of selected mixtures in PCB toxicity tests was a concern of USEPA in earlier comments 50 and 52 (above). Similarly, Westin et al. (1983) is also conservative. Westin et al. (1983) used just one treatment group, exposed the fish during a period of rapid growth, and found no effects. A concentration in fish associated with actual effects is not determined by this study, resulting in an unbounded NOAEL, a highly conservative representation of a TRV.
9	Appendix E	The table of contents shall be updated to reflect the additions of Sections 2.3.1.4 and 2.4.3.	The table of contents will be updated.

Note: Section, table, and figure numbers cited in comments dated March 15, 2012, and their respective responses, are those presented in the Draft Baseline Ecological Risk Assessment (BERA) dated March 15, 2012. Some of these numbers were subject to change when the Draft BERA was revised.

References:

Cooper, K.R., and M.L. Wintermyer, 2009. A critical review: 2,3,7,8-Tetrachlorodibenzo-*p*-dioxin (2,3,7,8-TCDD) effects on gonad development in bivalve mollusks. *Journal of Environmental Science and Health* Part C, Environmental, Carcinogenesis & Ecotoxicology Reviews 27(4):226-245.

Integral, 2010a. Technical Memorandum on Bioaccumulation Modeling, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA. September 2010.

Integral, 2010b. Sampling and Analysis Plan: Tissue Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

Integral, 2011. Field Sampling Report: Tissue Study, San Jacinto River Waste Pits Superfund Site. Prepared for McGinnes Industrial Maintenance Corporation, International Paper Company, and U.S. Environmental Protection Agency, Region 6. Integral Consulting Inc., Seattle, WA.

Miller, G., 2013. Personal Communication (e-mail to D. Keith, Anchor QEA LLC, regarding [comments on] San Jacinto RI and BERA, dated April 18, 2013). U.S. Environmental Protection Agency.

USEPA, 1988. Guidance for Conducting Remedial Investigations and Feasibility Studies under CERCLA. U.S. Environmental Protection Agency, Office of Emergency and Remedial Response, Washington, DC.

USEPA, 1993. Exposure Factors Handbook-Volume 1. General Factors. EPA/600/P-95/002Fa. U.S. Environmental Protection Agency, National Center for Environmental Assessment, Office of Research and Development, Washington, DC, and Versar Inc., Exposure Assessment Division, Springfield, VA.

Wintermyer, M.L., and K.R. Cooper, 2003. Dioxin/Furan and Polychlorinated Biphenyl Concentrations in Eastern Oyster (*Crassostrea virginica*, Gmelin) Tissues and the Effects on Egg Fertilization and Development. *J. Shellfish Res.* 22(3):737-746.